doi:10.7289/V5/TM-SWFSC-564

NOAA Technical Memorandum NMFS

JULY 2016

VIABILITY ASSESSMENT FOR PACIFIC SALMON AND STEELHEAD LISTED UNDER THE ENDANGERED SPECIES ACT: SOUTHWEST

Thomas H. Williams
Brian C. Spence
David A. Boughton
Rachel C. Johnson
Lisa G. Crozier
Nathan J. Mantua
Michael R. O'Farrell
Steven T. Lindley

NOAA-TM-NMFS-SWFSC-564

U.S. DEPARTMENT OF COMMERCE
National Oceanic and Atmospheric Administration
National Marine Fisheries Service
Southwest Fisheries Science Center

NOAA Technical Memorandum NMFS

This document should be cited as follows:

Williams, T.H., B.C. Spence, D.A. Boughton, R.C. Johnson, L.G. Crozier, N.J. Mantua, M.R. O'Farrell, and S.T. Lindley. 2016. Viability assessment for Pacific salmon and steelhead listed under the Endangered Species Act: Southwest. U.S. Department of Commerce, NOAA Technical Memorandum NMFS-SWFSC-564.

Individual sections (Recovery Domains) should be cited as follows:

Spence, B.C. 2016. North-Central California Coast Recovery Domain. Pages 32–82 *in* T.H. Williams, B.C. Spence, D.A. Boughton, R.C. Johnson, L.G. Crozier, N.J. Mantua, M.R. O'Farrell, and S.T. Lindley. Viability assessment for Pacific salmon and steelhead listed under the Endangered Species Act: Southwest. U.S. Department of Commerce, NOAA Technical Memorandum NMFS-SWFSC-564.

doi:10.7289/V5/TM-SWFSC-564

NOAA Technical Memorandum NMFS
This TM series is used for documentation and timely communication of preliminary results, interim reports, or special purpose information. The TMs have not received complete formal review, editorial control, or detailed editing.

JULY 2016

VIABILITY ASSESSMENT FOR PACIFIC SALMON AND STEELHEAD LISTED UNDER THE ENDANGERED SPECIES ACT: SOUTHWEST

Thomas H. Williams, Brian C. Spence, David A. Boughton, Rachel C. Johnson, Lisa G. Crozier*, Nathan J. Mantua, Michael R. O'Farrell, and Steven T. Lindley

NOAA National Marine Fisheries Service
SWFSC Fisheries Ecology Division
110 Shaffer Road
Santa Cruz, CA 95060

* NOAA National Marine Fisheries Service
Northwest Fisheries Science Center
2725 Montlake Boulevard E
Seattle, WA 98112

NOAA-TM-NMFS-SWFSC-564

U.S. DEPARTMENT OF COMMERCE
National Oceanic and Atmospheric Administration
National Marine Fisheries Service
Southwest Fisheries Science Center

(This page intentionally left blank)

Table of Contents

(This page intentionally blank)

List of Figures

Figure

List of Tables

Table

(This page intentionally blank)

Executive Summary

The Endangered Species Act (ESA) requires that the National Marine Fisheries Service (NMFS) review the status of listed species under its authority at least every five years and determine whether any species should be removed from the list or have its listing status changed. Assessments of the viability of ESA-listed Pacific salmonids were conducted by the Southwest Fisheries Science Center and the Northwest Fisheries Science Center. The information from these assessments is incorporated into the West Coast Region's status review, and the NMFS West Coast Region will make final determinations about any proposed changes in listing status, taking into account not only biological information (viability assessments) but also threats to the species and ongoing or planned protective efforts.

This report from the Southwest Fisheries Science Center covers 10 ESA-listed Evolutionarily Significant Units (ESUs) and Distinct Population Segments (DPSs) that lie wholly or partially in California. In this review, we consider 1) new information relevant to the delineation of ESU/DPS boundaries, and 2) new information on status and trends in abundance, productivity, spatial structure and diversity specifically addressed by viability criteria previously developed by Technical Recovery Teams (TRTs). These viability assessments summarize current information (through the 2014–2015 spawning year where available) with respect to the viability criteria developed by the TRTs. Consequently, the current assessments consider not only changes in populations that have occurred since the 2010 assessments but also the status of populations and ESUs/DPSs in relation to the viability criteria developed by the TRTs.

For eight of the ESUs/DPSs (Southern Oregon/Northern California Coho Salmon, Central California Coast Coho Salmon, California Coastal Chinook Salmon, Northern California Steelhead, Central California Coast Steelhead, South-central California Steelhead, Southern California Steelhead, and California Central Valley Steelhead) the new information suggests that there has been no change in extinction risk since 2010 viability assessments. For two ESUs (Central Valley Spring-run Chinook Salmon, and Sacramento River Winter-run Chinook Salmon) the new information suggests a change in extinction risk. The viability of Central Valley Spring-run Chinook salmon appears to have improved since the 2010 assessment, but this ESU is far from being viable and is still facing relatively high extinction risk. The viability of Sacramento River Winter-run Chinook Salmon has been reduced and the ESU faces greater extinction risk since the 2010 assessment.

(This page intentionally blank)

1 Introduction and Summary of Findings

1.1 Introduction

The Endangered Species Act (ESA) requires that the National Marine Fisheries Service (NMFS) review the status of listed species under its authority at least every five years and determine whether any species should be removed from the list or have its listing status changed. In June 2005, NMFS issued final listing determinations for 16 Evolutionarily Significant Units (ESUs) of Pacific Salmon (*Oncorhynchus* spp.), and in January 2006 NMFS issued final listing determinations for 10 Distinct Population Segments (DPS)s of steelhead (the anadromous form of rainbow trout: *O. mykiss*). The NMFS last conducted reviews in 2010. Therefore, the NMFS is conducting its 5-year status review of 28 currently listed Pacific salmonid ESUs/DPSs[1]. The review is being conducting by the NMFS West Coast Region Office. Assessments of the viability of ESA-listed Pacific salmonids are being conducted by the Southwest Fisheries Science Center and the Northwest Fisheries Science Center. The information from these assessments will be incorporated into the West Coast Region's review, and the NMFS West Coast Region will make final determinations about any proposed changes in listing status, taking into account not only biological information (viability assessments) but also threats to the species and ongoing or planned protective efforts.

This report covers 10 ESA-listed ESUs/DPSs that lie wholly or partially in California[2]. The Northwest Fisheries Science Center has developed a companion report for listed ESUs/DPSs in Oregon, Washington, and Idaho (Northwest Fisheries Science Center 2015).

In this review, we consider 1) new information relevant to the delineation of ESU/DPS boundaries, and 2) new information on status and trends in abundance, productivity, spatial structure and diversity specifically addressed by viability criteria previously developed by Technical Recovery Teams (TRTs). These viability assessments summarize current information (through the 2014–2015 spawning year where available) with respect to the viability criteria developed by the TRTs. Consequently, the current assessments consider not only changes in populations that have occurred since the 2010 assessments but also the status of populations and ESUs/DPSs in relation to the viability criteria developed by the TRTs.

[1] Federal Register Volume 80 Number 25, 6 February 2015, – see https://www.federalregister.gov/articles/2015/02/06/2015-02337/endangered-and-threatened-species-initiation-of-5-year-reviews-for-32-listed-species-of-pacific

[2] The Southern Oregon/Northern California Coast Coho Salmon ESU includes populations in coastal basins of southern Oregon.

1.2 An Overview of New Information for Consideration of Boundary Delineations of Listed California ESUs/DPSs

As previously discussed, NMFS is required to review the status of Endangered Species Act (ESA) listed groups every five years. As part of that process, it is necessary to evaluate the geographic or ecological boundaries of listed Evolutionarily Significant Units (ESUs) and Distinct Population Segments (DPSs) to determine if new information is available that suggests a boundary change could be warranted.

Chinook Salmon

Chinook salmon (*O. tshawytscha*) are distributed in coastal basins north of the Golden Gate (entrance to San Francisco Bay) and in the Sacramento/San Joaquin River and associated Bay/Delta systems of California's Central Valley. In California, six ESUs have currently been identified. The Southern Oregon/Northern California Coastal (SONCC) ESU includes populations from Cape Blanco in the north to the lower Klamath River in the south. The California Coastal (CC) ESU includes populations from Redwood Creek in the north to the Russian River (inclusive) in the south. The Upper Klamath and Trinity Rivers ESU includes populations spawning upstream of the confluence of these two rivers. The Central Valley contains three ESUs, one of which, Fall-run/Late Fall-run Chinook salmon, currently extends from Carquinez Strait into the Sacramento and San Joaquin rivers and their tributaries. The other two ESUs, Sacramento River Winter-run Chinook salmon (SRWRC) and Central Valley Spring-run Chinook salmon (CVSRC), do not extend into the Bay/Delta Region at all. The Coastal California Chinook Salmon ESU and the Central Valley Spring-run Chinook Salmon ESU are ESA Threatened, the Sacramento River Winter-run Chinook Salmon ESU is ESA Endangered, and the other ESUs are not listed.

The previous viability assessment (Williams et al. 2011) discussed the fact that populations that lie between the lower boundary of the Central Valley Fall-run ESU (Carquinez Straits) and the southern boundary of the California Coastal ESU (Russian River) were not included in either ESU, despite the fact that Chinook salmon had been reported in several basins. Available genetic evidence indicated fish from the Guadalupe and Napa rivers in San Francisco and San Pablo bays had close affinity with Central Valley Fall-run Chinook (C. Garza, NMFS SWFSC, unpublished data; Garza and Pearse 2008), and it was recommended that fish from these two watersheds be included in the Central Valley Fall-run Chinook Salmon ESU. Evidence for fish in Lagunitas Creek was equivocal, with 17 samples assigned almost equally between California Coastal Chinook Salmon and Central Valley Fall-run Chinook Salmon. The review team tentatively concluded that Lagunitas Creek Chinook salmon should be considered part of the California Coastal ESU pending additional data (Williams et al. 2011). NMFS subsequently indicated that a boundary change was under consideration (76 FR 50447); however, no action has been taken to date. There is no new genetic information that helps resolve this issue (C. Garza, NMFS SWFSC, personal communication).

The San Joaquin Delta and entire watershed is excluded as Critical Habitat and its populations are considered extirpated (64 FR 50394; 70 FR 52488). Information on the presence of fish exhibiting spring-run behavior in San Joaquin River tributaries may represent passive recolonization of Central Valley Spring-run Chinook salmon in the San Joaquin River basin. Thus, there is value in continuing to monitor these populations to evaluate the extent to which populations in the San Joaquin River tributaries warrant inclusion in the ESU boundary. No new information suggests that the boundary of this ESU should change or that its status as an ESU should change.

Coho Salmon

Coho salmon (*O. kisutch*) are distributed in coastal California basins from the Oregon border in the north to Monterey Bay in the south and historically were present in the San Francisco/San Pablo Bay system, where they are now extirpated. Populations to the north of Punta Gorda, from the Mattole River north, are assigned to the SONCC Coho Salmon ESU, whereas populations to the south of Punta Gorda to Aptos Creek are part of the Central California Coast (CCC) Coho Salmon ESU. The SONCC Coho Salmon ESU is ESA Threatened, whereas the CCC Coho Salmon ESU is ESA Endangered. In 2003, NMFS Southwest Fisheries Science Center conducted an extensive genetic survey of coho salmon populations in coastal California. Genetic samples were taken from juvenile coho salmon collected at 30 sites in 23 different watersheds spanning the SONCC- and CCC-Coho Salmon ESUs. Multiple analyses of microsatellite data provided consistent and strong support for the current ESU boundary at Punta Gorda (Gilbert-Horvath et al., in press). These data show clear separation between populations north and south of Punta Gorda, the current southern boundary of the ESU. The Biological Review Team for the Oregon Coast Coho Salmon ESU reviewed genetic data and concluded that a reconsideration of the ESU boundary between the SONCC Coho Salmon ESU and Oregon Coast Coho Salmon ESU was not necessary (Stout et al. 2010).

The initial status review for the CCC-Coho Salmon ESU (Weitkamp et al. 1995) defined the ESU as populations from Punta Gorda southward to and including the San Lorenzo River. Since that time, the boundary has been extended southward to include Soquel and Aptos creeks (77 FR 19552) based on analysis of historical and recent evidence of occurrence as well as environmental conditions in these two watersheds (Spence et al. 2011). Successful reproduction of coho salmon in Soquel Creek was again documented in the summer of 2015 (B. Spence and J. Kiernan, NMFS SWFSC, personal communication), which supports the boundary extension.

Steelhead

Steelhead/rainbow trout (*O. mykiss*) are distributed throughout California, in coastal streams from the Oregon border in the north to the border with Mexico in the south, and throughout the Central Valley. In addition, *O. mykiss* populations are present in nearly all of the tributaries upstream of dams constructed over the last century. There are a total of six DPSs in California, with one in the Central Valley and five on the coast.

The Klamath Mountains Province Steelhead DPS begins at the Elk River in Oregon and extends to the Klamath/Trinity basin in California, inclusive. The Northern California Steelhead DPS extends from Redwood Creek in the north to the Gualala River in the south, inclusive. The Central California Coast Steelhead DPS begins at the Russian River, contains populations in streams tributary to the San Francisco/San Pablo Bay system, and stretches south to Aptos Creek, inclusive. The South-Central California Coast Steelhead DPS starts at the Pajaro River in the Monterey Bay Region and continues to Arroyo Grande in San Luis Obispo Bay. The Southern California Steelhead DPS begins at the Santa Maria River, inclusive, and stretches to the border with Mexico. The California Central Valley Steelhead DPS includes all populations in the Sacramento/San Joaquin River system and its delta. All of these DPSs include only potentially anadromous fish below definitive natural or manmade barriers to anadromy. The Klamath Mountains Province DPS is not ESA listed, the Southern California DPS is ESA Endangered, and all of the others are ESA Threatened.

We recommend a change in boundary delineation of the California Central Valley Steelhead DPS. This DPS includes steelhead populations spawning in the Sacramento and San Joaquin rivers and their tributaries. Hatchery stocks within the DPS include Coleman National Fish Hatchery and Feather River Hatchery; steelhead in the Nimbus Hatchery and Mokelumne River Hatchery are currently excluded from the DPS. New genetic analysis show that the steelhead broodstock currently propagated in the Mokelumne River Hatchery is genetically similar to the steelhead broodstock in the Feather River Hatchery (Pearse and Garza 2015), which is consistent with documentation on the recent transfers of eggs from the Feather River Hatchery for broodstock at the Mokelumne River Hatchery. The Nimbus Hatchery steelhead remain genetically divergent from the California Central Valley DPS lineages, consistent with their founding from coastal steelhead populations, and remain excluded from the DPS (Pearse and Garza 2015). Thus, we recommend a change in boundary delineation; the boundary of the California Central Valley DPS should be modified to include steelhead from the Mokelumne River Hatchery.

In the previous viability assessment (Williams et al. 2011) it was determined that new genetic population structure data not available at the time of the original DPS delineation suggest several potential boundary changes may be warranted for coastal California DPSs. Williams et al. (2011) discuss these data and potential boundary delineation changes. For example, Clemento et al. (2009) found no evidence for a genetic boundary between the two southernmost DPSs, and Bjorkstedt et al. (2005) presented analyses indicating that genetic boundaries in the northern coastal DPSs coincide with current boundaries in one regional area, between the Northern California and Central California Coast DPSs. No potential changes in DPS boundaries involving the Central Valley were suggested by these recent genetic data. Since the previous assessment, data analyzed by Bjorkstedt et al. (2005) has been published (Garza et al. 2014).

Based on these new data and information, it was recommended that a Biological Review Team (BRT) be convened to compile, review, and evaluate the best available scientific and commercial information on steelhead genetics, life history and biology, and the ecological/habitat requirements of steelhead that are relevant to evaluate current boundaries and potential DPS boundary changes. The BRT review has yet to be conducted, and therefore the existing boundary delineations of coastal California steelhead DPSs were used in this report.

1.3 Summary of Findings

Climatic conditions affect salmonid abundance, productivity, spatial structure, and diversity through direct and indirect impacts at all life stages (e.g., Crozier et al. 2008; ISAB 2007; Lindley et al. 2007; Moyle et al. 2013; Wainwright and Weitkamp 2013). Salmon have adapted to a wide variety of climatic conditions in the past, and thus inherently could likely survive substantial climate change at the species level in the absence of other anthropogenic stressors.

Currently, the adaptive ability of these threatened and endangered species is depressed due to reductions in population size, habitat quantity and diversity, and loss of behavioral and genetic variation. Without these natural sources of resilience, systematic changes in local and regional climatic conditions due to anthropogenic global climate change will likely reduce long-term viability and sustainability of populations in many of these ESUs and DPSs. Adapting to climate change may eventually involve changes in multiple life-history traits and/or local distribution, and some populations or life-history variants might die out. Importantly, the character and magnitude of these effects will vary within and among ESUs/DPSs.

California has experienced well below average precipitation in each of the past four water years (2012, 2013, 2014, and 2015), record high surface air temperatures the past two water years (2014 and 2015), and record low snowpack in 2015. Some paleoclimate reconstructions suggest that the current four-year drought is the most extreme in the past 500 or perhaps more than 1000 years. Anomalously high surface temperatures have made this a “hot drought”, in which high surface temperatures substantially amplified annual water deficits during the period of below average precipitation.

Four consecutive years of drought and the past two years of exceptionally high air, stream, and upper-ocean temperatures have together likely had negative impacts on the freshwater, estuary, and marine phases for many populations of Chinook salmon, coho salmon, and steelhead.

Monitoring in coastal basins in California, most notably those basins north of Aptos Creek (Santa Cruz County) to the Oregon border, has improved considerably since the 2010 viability assessment as a result of the implementation of the Coastal Monitoring Plan[3]. The Coastal Monitoring Plan framework provides population abundance estimates at the appropriate spatial scale (i.e., population unit) and has greatly expanded the number of populations being monitoring in these coastal basins. Unfortunately, lack or limited implementation of the Coastal Monitoring Plan in the South Central California Coast and

[3] For information on the California Coastal Monitoring Program: http://www.calfish.org/ProgramsData/ConservationandManagement/CaliforniaCoastalMonitoring.aspx

Southern California Coast Steelhead DPSs prevents an adequate assessment of viability of these steelhead DPSs. In addition, the few estimates available at the population unit spatial scale from the Oregon portion of the Southern Oregon/Northern California Coho Salmon ESU for the 2010 assessment are no longer collected and therefore no estimates at the population spatial scale are available for Oregon populations of the SONCC-Coho Salmon ESU.

For eight of the ESUs/DPSs (Southern Oregon/Northern California Coho Salmon, Central California Coast Coho Salmon, California Coastal Chinook Salmon, Northern California Steelhead, Central California Coast Steelhead, South-central California Steelhead, Southern California Steelhead, and California Central Valley Steelhead) the new information suggests that there has been no change in extinction risk since 2010 viability assessment (Table 1.1). For two ESUs (Central Valley Spring-run Chinook Salmon, and Sacramento River Winter-run Chinook Salmon) the new information suggests a change in extinction risk. The viability of Central Valley Spring-run Chinook salmon appears to have improved since the 2010 assessment, but this ESU is far from being viable and is still facing relatively high extinction risk. The viability of Sacramento River Winter-run Chinook Salmon has been reduced and the ESU faces greater extinction risk since the 2010 assessment.

For considering if there has been a change in the extinction risk of an ESU/DPS, data and information were considered in the context of the Technical Recovery Team (TRT) viability criteria and not solely determined by a change in trend/viability since the 2010 assessment. In general, as Table 1.1 illustrates, ESUs or DPSs that the previous assessment (Williams et al. 2011) considered likely to become endangered are missing populations from diversity strata and only a portion of the populations are currently known to be extant.

Table 1.1. Summary of current listing status, and summary of current assessments of new and additional data, changes in trends/viability since 2010 assessment, spatial extent of current populations, and current viability of populations. Note that known low-risk independent populations are those populations that are demonstrably low-risk.

		Current review				
ESU/DPS	Listing status	Change in trend/viability since 2010 assessment	Diversity strata occupied (occupied/total)	Extant independent populations (extant/total)	Known low-risk independent populations (viable/total)	5-year update (this assessment) Change in extinction risk
SONCC coho salmon	Threatened	Mixed	7/7	27/27	0/27	No change
CCC coho salmon	Endangered	Mixed	4/5	8/12[a]	0/12	No change
CA Coastal Chinook salmon	Threatened	Mixed	4/4 fall run[b] 0/2 spring run	14/15 fall run[c] 0/6 spring run	0/15 0/6	No change
Northern CA steelhead	Threatened	Mixed	5/5 winter run 2/2 summer run	42/42 5/10	0/42 0/10	No change
CCC steelhead	Threatened	Uncertain	5/5	30?/37[d]	0/37	No change
South-central CA steelhead	Threatened	Negative	5/5	12?/13	0/13	No change
Southern CA steelhead	Endangered	Uncertain	3?/5	11?/18	0/18	No change
CV Spring Chinook salmon	Threatened	Improved	3/4 (2/4)[e]	4/18	1/18	Decreased risk of extinction
CA Central Valley steelhead	Threatened	Uncertain		50?/81[f]	0/81[g]	No change
Sac. River Winter Chinook salmon	Endangered	Negative	0/1	1/4	1/4	Increased risk of extinction

a – Coho salmon have been occasionally observed in Walker Creek, Pescadero Creek, and the San Lorenzo River in the last 10 years; however, all of these observations are apparently a consequence of hatchery strays, and there is no evidence of persistent occurrence in these three watersheds.

b – Populations in the North-Central Coastal Diversity Stratum – were previously believed extirpated; however, recent monitoring has indicated small numbers of spawners in the Ten Mile River, Noyo River, and Big River.

c – Five populations were previously thought to be extirpated (Ten Mile, Noyo, Big, Navarro, and Garcia rivers); however, recent monitoring indicates low numbers of spawners are returning in most years.

d - Current occupancy is uncertain for 7 populations in San Francisco Bay area and coastal Marin and Santa Cruz counties; in some of these systems, *O. mykiss* may still be present, but it is unclear if the anadromous form still occurs in the watershed.

e – As proposed by the TRT, one diversity stratum for Central Valley Spring-run Chinook salmon consisted only of dependent populations, so only two of four diversity strata in this ESU are occupied by independent populations.

f – Populations assumed extant if some historical habitat currently accessible.

g – Most populations in this DPS are data deficient; the few with data are at high risk of extinction.

2 Climate and Ocean Conditions

Lisa G. Crozier and Nathan J. Mantua

2.1 Climate Effects

Projected Impacts of Future Climate Change on West Coast Salmon

Climatic conditions affect salmonid abundance, productivity, spatial structure, and diversity through direct and indirect impacts at all life stages (e.g., ISAB 2007; Lindley et al. 2007; Crozier et al. 2008; Moyle et al. 2013; Wainwright and Weitkamp 2013). Salmon have adapted to a wide variety of climatic conditions in the past, and thus inherently could likely survive substantial climate change at the species level in the absence of other anthropogenic stressors.

Currently, the adaptive ability of these threatened and endangered species is depressed due to reductions in population size, habitat quantity and diversity, and loss of behavioral and genetic variation. Without these natural sources of resilience, systematic changes in local and regional climatic conditions due to anthropogenic global climate change will likely reduce long-term viability and sustainability of populations in many of these ESUs and DPSs. Adapting to climate change may eventually involve changes in multiple life-history traits and/or local distribution, and some populations or life-history variants might not survive. Importantly, the character and magnitude of these effects will vary within and among ESUs/DPSs.

The Intergovernmental Panel on Climate Change (IPCC) and U.S. Global Change Research Program recently published updated assessments of anthropogenic influence on climate, as well as projections of climate change over the next century (IPCC 2013; Melillo et al. 2014). Reports from both groups document ever-increasing evidence that recent warming bears the signature of rising concentrations of greenhouse gas emissions.

The U.S. Global Change Research Program report contains regional-focus chapters for the northwest (Snover et al. 2013; Mote et al. 2014) and southwest U.S. (Garfin et al. 2014). These regional reports synthesize information from an extensive literature review, including a broad array of analyses of regional observations and climate change projections. These synthesis reports were the primary source for this West Coast summary. References to the primary literature can be found in those reports. Updates to this summary can be found in annual literature reviews conducted by National Marine Fisheries Service (available at http://www.nwfsc.noaa.gov/trt/lcm/freshwater_habitat.cfm).

Historical Climate Trends

Observed historical trends in climate reflect the early influence of greenhouse gas emissions, and often indicate the general direction of future climate change. These observations also reflect natural variability in climate at multiple time scales. Natural variability alternately intensifies and relaxes (or partially reverses) the long-term trends. Attribution of historical trends to anthropogenic factors is most certain at the global scale over time scales of centuries to millennia because at these scales we can better account for natural variability.

Historical records show pronounced warming in both sea-surface and land-based air temperatures. There is moderate certainty that the 30-year average temperature in the Northern Hemisphere is now higher than it has been over the past 1,400 years. In addition, there is high certainty that ocean acidity has increased with a drop in pH of 0.1. Furthermore, glaciers and sea-ice have receded, while sea level has risen (global mean rose 0.19 m over the last century). In recent decades, the frequency of extreme high temperature or heavy precipitation events has increased in many regions. An anthropogenic influence on this shift in frequency is "very likely" (IPCC 2014).

Regional and local trends include the following observations:

- In both the Northwest and Southwest:
    - air temperatures have increased since the late 1800s
    - springtime snow-water equivalent has decreased (since 1950)
    - snowmelt occurs earlier in the year
- In the Southwest, drought over the past four years is unprecedented in the historical record and may be the worst in over 1,000 years. This drought has been attributed to a combination of anthropogenic influence on temperature and natural variability in precipitation (Williams et al. 2015). Trends in precipitation vary spatially up or down, with no statistically significant trends in precipitation averages or extremes in the Northwest.
- In both the Northwest and Southwest, widespread tree mortality has been observed, wildfires have increased in both frequency and area burned, and insect outbreaks have increased (Garfin et al. 2014; Mote et al. 2014).
- Historical trends in the California Current are heavily influenced by patterns in wind-driven ocean circulation, which correlates with large-scale climate drivers such as the North Pacific Gyre Oscillation (Peterson et al. 2013) and Pacific Decadal Oscillation (Jacox et al. 2014). Spatially variable trends in upwelling intensity (Jacox et al. 2014) and hypoxia (Peterson et al. 2013), and longer trends in atmospheric forcing and sea surface temperature (Johnstone and Mantua 2014) probably reflect natural climate variability to a much greater extent than anthropogenic forcing.
- The pH of the California Current has decreased by about 0.1 and by 0.5 in aragonite saturation state since pre-industrial times (Hauri et al. 2009). Furthermore, infrastructure in coastal areas is increasingly damaged by erosion and flooding (Garfin et al. 2014; Mote et al. 2014; Sweet et al. 2014).

Projected Climate Changes

Trends in warming and ocean acidification are highly likely to continue during the next century (IPCC 2103). Scenarios considered in the IPCC fifth assessment report range from the severely curtailed greenhouse gas emissions of representative concentration pathway (RCP) 2.6 to business as usual in RCP 8.5.

Based on means across global climate models spanning the full breadth of these emissions scenarios, IPCC projected the following ranges across the Northern Hemisphere by 2081-2100:

- Spring snow cover declines of 7-25%
- Glacier recessions of 15-85%
- Sea surface temperature increases of 1.1-3.6°C
- Global sea level increases of 11-38 inches
- Global ocean pH decreases of 38 to 109%, which correspond to a drop in pH of 0.14-0.32.

Regional projections add spatial variability and specificity to these themes. In winter across the west, the highest elevations (e.g., in the Rocky Mountains) will shift from consistent longer (>5 months) snow-dominated winters to a shorter period (3-4 months) of reliable snowfall (Klos et al. 2014); lower, more coastal or more southerly watersheds will shift from consistent snowfall over winter to alternating periods of snow and rain ("transitional"); lower elevations or warmer watersheds will lose snowfall completely, and rain-dominated watersheds will experience more intense precipitation events and possible shifts in the timing of the most intense rainfall (e.g., Salathé et al. 2014).

By the 2080s, Tohver et al. (2014) anticipate a complete loss of snow-dominated basins in the Cascades and U.S. portion of the Rockies, with only a few "mixed" basins of rain- and snow-fed runoff remaining at the highest elevations. Flooding is projected to increase in basins that experience a mix of snow and rain in winter (Mote et al. 2014; Salathé et al. 2014; Tohver et al. 2014). Erosion and flooding in coastal areas are projected to increase with rising sea levels (Garfin et al. 2014; Mote et al. 2014; Sweet et al. 2014).

Among seasons, the greatest temperature shifts are expected in summer. Warmer summer air temperatures will increase both evaporation and direct radiative heating. When combined with reduced winter water storage, warmer summer air temperatures will lead to lower minimum flows in many watersheds. Higher summer air temperatures will depress minimum flows and raise maximum stream temperatures even if annual precipitation levels do not change (e.g., Sawaske and Freyberg 2014). Summer precipitation also influences summer flows, but projections for precipitation are less certain than for temperature. Coastal weather can differ from region-wide projections due to changes in fog, on-shore winds, or precipitation (Johnstone and Dawson 2010; Potter 2014).

Widespread ecosystem shifts are very likely, and may be abrupt due to disturbances from increasing wildfires, insect outbreaks, droughts, and tree diseases (Garfin et al. 2014; Mote et al. 2014). Climate projections often favor invasive fish species over native species, with declines exacerbated by the greater vulnerability of native species to existing anthropogenic stressors (Lawrence et al. 2012; Lawrence et al. 2014; Quiñones and Moyle 2014).

In response to projected changes in both climate and land use practices, estuary dynamics are expected to change as well, with depth and salinity altered by changing sea level, upwelling regimes, and freshwater input (Yang et al. 2015). Intense upwelling events can move hypoxic and acidic water into estuaries, especially when freshwater input is reduced (e.g., Columbia River estuary, Roegner et al. 2011). Sea level projections differ at local versus global scales due to local wind and temperature trends and land movement. Specifically, the National Research Council (2012) predicted a lower rise in sea level off the coasts of Washington and Oregon (62 cm) than off the coast of California (92 cm) by 2100.

Higher sea surface temperatures and increased ocean acidity are predicted for marine environments in general (IPCC 2013). However, regional marine impacts will vary, especially in relation to productivity. The California Current is strongly influenced by seasonal upwelling of cool, deep, water that is high in nutrients and low in dissolved oxygen and pH. Ecological effects of climate change in the California Current are very sensitive to impacts on upwelling intensity, timing, and duration. Projections of how climate change will affect upwelling are highly variable across models, with predicted trends ranging from negative to positive (Bakun 1990; Mote and Mantua 2002; Snyder et al. 2003; Diffenbaugh et al. 2008; Bakun et al. 2010). An analysis of 21 global climate models found that most predicted a slight decrease in upwelling in the California Current, although there is a latitudinal cline in the strength of this effect, with less impact toward the north (Rykaczewski et al. 2015).

Much of the near-shore California Current is expected to be corrosive (undersaturated in aragonite) in the top 60 m during all summer months within the next 30 years, and year-round within 60 years (Gruber et al. 2012). Thermal stratification and hypoxia are expected to increase (Doney et al. 2014).

Impacts on Salmon

Studies examining the effects of long-term climate change to salmon populations have identified a number of common mechanisms by which climate variation is likely to influence salmon sustainability. These include direct effects of temperature such as mortality from heat stress, changes in growth and development rates, and disease resistance. Changes in the flow regime (especially flooding and low flow events) also affect survival and behavior. Expected behavioral responses include shifts in seasonal timing of important life-history events, such as the adult migration, spawn timing, fry emergence timing, and juvenile migration.

Indirect effects on salmon mortality, growth rates and movement behavior are also expected to follow from changes in the freshwater habitat structure and the invertebrate and vertebrate community, which governs food supply and predation risk (Petersen and Kitchell 2001; ISAB 2007; Crozier et al. 2008). Both direct and indirect effects of climate change will vary among Pacific salmon ESUs and among populations in the same ESU. Adaptive change in any salmonid population will depend on the local consequences of climate change as well as ESU-specific characteristics and existing local habitat characteristics.

Because climate has such profound effects on survival and fecundity, salmon physiology and behavior are exquisitely adapted to local environmental conditions. These adaptations vary systematically among populations and are exhibited in traits such as age and timing of juvenile and adult migrations, with potential differences in physiology and migration routes (Quinn 2005). These traits often have a significant plastic (non-genetic) component, which allows them to respond quickly to environmental change. Yet these traits also differ genetically among populations (Carlson and Seamons 2008).

Directional climate change could therefore drive many salmonid populations into a maladaptive state. Such an outcome would likely cause reductions in abundance, productivity, population spatial structure, and population diversity. In some cases, this can lead to extirpation if a population cannot adapt quickly enough. In other cases an adaptive solution may not exist because of conflicting pressures within or between life stages.

Climate impacts in one life stage generally affect body size or timing in the next life stage. For this reason, the cumulative life-cycle effects of climate change must be considered to fully appreciate the scope of risk to a given population. Even without interactions among life stages, the sum of impacts in many stages will have cumulative effects on population dynamics.

Climate effects tend to be negative across multiple life stages (Healey 2011; Wade et al. 2013; Wainwright and Weitkamp 2013). However, there may be mitigating responses in some ESUs or life stages. Individualistic impacts within and among ESUs will depend on factors such as existing physical and biological heterogeneity, proximity to the limits of physiological tolerance under present climate conditions, and the extent of localized climate change.

In many cases, directional climate change exacerbates existing anthropogenic threats. Examples include streams or rivers where stream temperatures are already elevated due to land-use modifications (Battin et al. 2007) or where flow is reduced due to water diversions (Walters et al. 2013). In the Columbia River, dams have altered the hydrological regime by causing an earlier and smaller freshet, which is the same type of effect expected from climate change (Naik and Jay 2011a; Naik and Jay 2011b). Any of these stressors in combination with one another or with climate impacts will present pressures of much greater concern than they would individually, but they also offer potential solutions (McClure et al. 2013).

Changes in winter precipitation will likely affect incubation and/or rearing stages of most populations. Changes in the intensity of cool-season precipitation could influence migration cues for fall and spring adult migrants, such as coho salmon and steelhead. Egg survival rates may suffer from more intense flooding that scours or buries redds.

Changes in hydrological regime, such as a shift from mostly snow to more rain, could drive changes in life history, potentially threatening diversity within an ESU. It is possible that even characteristic life-history traits used to help define the ESU will be threatened. For example, the juvenile freshwater rearing period is very sensitive to temperature, with the yearling life-history strategy used only by populations in cooler watersheds (Beechie et al. 2006). Frequency of the yearling life-history type will likely decline as movement downstream into estuaries or near-shore habitat is initiated at younger ages. Implications of this behavioral shift for juvenile survival, ocean migration behavior, and age at maturity are uncertain.

Changes in summer temperature and flow will affect both juvenile and adult stages in some populations, especially those with yearling life histories and summer migration patterns. Juvenile rearing and migration survival is often correlated with these factors (Quinn 2005; Crozier and Zabel 2006; Crozier et al. 2010).

Adults that migrate or hold during peak summer temperatures can experience very high mortality in unusually warm years. For example, in 2015 only 4% of adult Redfish Lake sockeye salmon survived the migration from Bonneville Dam to Lower Granite Dam after confronting temperatures over 22°C in the lower Columbia River. After prolonged exposure to temperatures over 20°C, salmon are especially likely to succumb to diseases that they might otherwise have survived (Materna 2001; Miller et al. 2014). They are also more vulnerable to any sort of stress, such as catch-and-release fisheries (Boyd et al. 2010).

Changing hydrology and temperature will also affect the timing of smolt migrations and spawning (Crozier and Hutchings 2014; Hayes et al. 2014; Otero et al. 2014). If smolts migrate at a smaller size because they leave freshwater habitat earlier, they might have lower survival due to size-selective predation (Thompson and Beauchamp 2014). Marine arrival timing is extremely important for smolt-to-adult survival (Scheuerell et al. 2009), and has been historically synchronized with the timing and predictability of favorable ocean conditions (Spence and Hall 2010). Given the uncertain effects of climate change on upwelling timing and intensity, impacts on juvenile survival from shifts in migration timing are also difficult to predict.

In some populations, behavior during the early ocean stage is consistent among years, suggesting a genetic rather than a plastic response to environmental conditions (Burke et al. 2014; Hassrick et al. 2016). These populations might change their behavior over time if the fitness landscape changes, but responses will likely be relatively slow and could be dominated by decadal ocean dynamics or productivity outside the California Current (e.g., the Gulf of Alaska for northern migrants).

Other populations show more variable behavior after ocean entry (Weitkamp 2010; Fisher et al. 2014), and some show heightened sensitivity to interannual climate variation, such as the El Niño Southern Oscillation (L. Weitkamp, NMFS NWFSC, personal communication). Such variability might increase ESU-level resilience to climate change, assuming some habitats remain highly productive.

Marine migration patterns could also be affected by climate-induced contraction of thermally suitable habitat. Abdul-Aziz et al. (2011) modeled changes in summer thermal ranges in the open ocean for Pacific salmon under multiple IPCC warming scenarios. For chum salmon, pink salmon, coho salmon, sockeye salmon, and steelhead, they predicted contractions in suitable marine habitat of 30-50% by the 2080s, with an even larger contraction (86-88%) for Chinook salmon under the medium and high emissions scenarios (A1B and A2).

Northward range shifts are a climate response expected in many marine species, including salmon (Cheung et al. 2015). However, salmon populations are strongly differentiated in the northward extent of their ocean migration, and hence will likely respond individualistically to widespread changes in sea surface temperature.

In most Pacific salmon species, size at maturation has declined over the past several decades. This trend has been attributed in part to rising sea surface temperatures (Bigler et al. 1996; Pyper and Peterman 1999; Morita et al. 2005). Mechanisms involved in such responses are likely complex, but appear to reflect a combination of density-dependent processes, including increased competition due to higher salmon abundance in recent years and temperature (Pyper and Peterman 1999). Temperature-related size effects could involve increased metabolic costs at higher temperatures, and/or shifts in spatial distribution in response to ocean conditions. Younger spawners affect population growth rates by exhibiting lower fecundity and reducing the population stability that stems from having multiple age classes reproduce.

Numerous researchers have reported that salmon marine survival is highly variable over time and often correlated with large-scale climate indices (Mueter et al. 2002; Mueter et al. 2005; Petrosky and Schaller 2010; Litzow et al. 2014; Stachura et al. 2014; Sydeman et al. 2014). For example, Pacific salmon from Washington and Oregon exhibited extremely low marine survival and dramatic population declines during a "warm phase" of the Pacific Decadal Oscillation in the 1980s and 1990s (Levin 2003; Zabel et al. 2006). These declines were attributed to low ocean productivity in the warm ocean of that period.

Many fish communities, including key salmon prey and predators, experience changes in abundance and distribution during warm ocean periods (Pearcy 2002; Wing 2006; Cheung et al. 2009). However, food chain dynamics in the open ocean are flexible and difficult to predict into the future.

The full implications of ocean acidification on salmon are not known at this time. Olfaction and predator-avoidance behavior are negatively affected in some fish species, including pink salmon (Leduc et al. 2013; Ou et al. 2015). Pink salmon also showed reductions in growth and metabolic capacity under elevated CO_2 conditions (Ou et al. 2015). Some high-quality salmon prey (e.g., krill) might be negatively affected by ocean acidification, but there are several possible pathways by which higher trophic levels might compensate for changes at a lower trophic level. From their analysis of multi-trophic responses to ocean acidification, Busch et al. (2013) concluded that impacts to salmon could conceivably be positive. However, they emphasized that a better understanding of both direct and indirect feedback loops is necessary before drawing definitive conclusions.

To what extent a future warmer ocean will mimic historical conditions of warm-ocean, low-survival periods is not known. Current indications are that a warmer Pacific Ocean is generally less productive at mid latitudes, and hence likely to be less favorable for salmon.

Analysis of ESU-specific vulnerabilities to climate change by life stage will be available in the near future, upon completion of the *West Coast Salmon Climate Vulnerability Assessment* being conducted by the National Marine Fisheries Service. Climate effects on one Pacific salmon ESU, the Oregon coastal coho salmon ESU, were recently assessed by Wainwright and Weitkamp (2013); many of the effects they reported for this ESU are likely shared by other ESUs (Table 2.1).

In summary, both freshwater and marine productivity tend to be lower in warmer years for most populations considered in this assessment. These trends suggest that many populations might decline as mean temperature rises. However, the once historically high abundance of many California populations of Pacific salmonids is reason for optimism and warrants considerable effort to restore the natural climate resilience of these species.

2.2 2012-2015 Drought Impacts on West Coast Salmon and Salmon Habitat

California has experienced well below average precipitation in each of the past four water years (2012, 2013, 2014, and 2015), record high surface air temperatures the past two water years (2014 and 2015), and record low snowpack in 2015. Some paleoclimate reconstructions suggest that the current four-year drought is the most extreme in the past 500 or perhaps more than 1000 years. Anomalously high surface temperatures have made this a "hot drought", in which high surface temperatures substantially amplified annual water deficits during the period of below average precipitation.

The combination of four consecutive years of drought and record-high air temperatures in 2014 and 2015 favored elevated stream temperatures, and these were documented to have severe impacts in some watersheds. The lack of cold water behind Shasta Dam on the upper Sacramento River, in combination with water release decisions, led to unfavorably high stream temperatures below Shasta Dam 2014 (SRTTG 2014) and 2015. Brood years 2014 and 2015 experienced the lowest egg-to-fry survival rates on record (5.6% and 4.5%, respectively) (Poytress 2016, PFMC 2016). Concerns over a high potential for fish kills in the Klamath River basin were also high in the summers of 2014 and 2015 because of warm stream temperatures and elevated presence of pests and pathogens detected in salmon. These concerns prompted emergency reservoir releases aimed at lowering downstream temperatures to alleviate those risks.

Exceptionally Warm Ocean Conditions in the NE Pacific

Much of the northeast Pacific Ocean, including parts typically used by California salmon and steelhead, experienced exceptionally high temperatures of the upper 100 m of the ocean beginning early in 2014 and areas of extremely high ocean temperatures continued to cover most of the northeast Pacific Ocean through all of 2015 (NMFS 2015). A "warm blob" formed offshore of the Pacific Northwest (PNW) region in fall 2013 (Bond et al. 2015). Off the coast of southern and Baja California, upper ocean temperatures became anomalously warm in spring 2014, and this warming spread to the central California coast in July 2014. In fall 2014, a shift in wind and ocean current patterns caused the entire northeast Pacific Ocean domain to experience unusually warm upper ocean temperatures from the West Coast offshore for several hundred kilometers. In spring 2015 nearshore waters from Vancouver Island south to San Francisco mostly experienced

Table 2.1. Projected climate changes affecting Oregon coho salmon (*O. kisutch*), as reported by Wainwright and Weitkamp (2013). Abbreviations: LWD (large woody debris) – – strongly negative, – negative, ○ neutral, + positive, + + strongly positive.

Physical/chemical pattern	Certainty of change	Process affecting Oregon coast coho salmon	Range of effects					Certainty of effect
			--	-	○	+	++	
Terrestrial habitat								
Warmer, drier summers	Moderate	Increased fires, increased tree stress and disease affect LWD, sediment supplies, riparian zone structure	X	X	X			Low
Reduced snow pack, warmer winters	High	Increased growth of higher elevation forests affect LWD, sediment, riparian zone structure			X	X		Low
Freshwater habitat								
Reduced summer flow	High	Less accessible summer rearing habitat		X				Moderate
Earlier peak flow	High	Potential migration timing mismatch	X	X	X			Moderate
Increased floods	Moderate	Redd disruption, juvenile displacement, sediment dynamics	X	X	X	X		Moderate
Higher summer stream temperature	Moderate	Thermal stress, restricted habitat availability, increased susceptibility to disease, parasites, and predators	X	X				Moderate
Higher winter stream temperatures	Low	Increased fry growth, shorter incubation				X	X	Low
Estuarine habitat								
High sea level	High	Reduced availability of wetland habitats	X	X				Moderate
Higher water temperature	Moderate	Thermal stress, increased susceptibility to disease, parasites, and predators	X	X				Moderate
Combined effects		Changing ecosystem composition and structure	X	X	X	X	X	Low
Marine habitat								
Higher ocean temperature	High	Thermal stress, shifts in migration, range shifts, susceptibility to disease, parasites, and predators	X	X				Moderate
Intensified upwelling	Moderate	Increased nutrients (food supply), coastal cooling, ecosystem shifts; increased offshore transport			X	X	X	Low
Delayed spring transition	Low	Food timing mismatch with juvenile migrants, ecosystems shifts		X	X			Low
Intensified stratification	Moderate	Reduced food supply, change in habitat structure	X	X				Low
Increased acidity	High	Disruption of food supply, ecosystem shifts	X	X				Moderate
Combined effects		Changing ecosystem composition and structure; food supply and predation		X	X	X	X	Low

strong and at times above average coastal upwelling that created a relatively narrow band (~50 to 100 km wide) of near normal upper ocean temperatures, while the exceptionally high temperature waters remained offshore and in coastal regions to the south and north.

Expectations for Future Climate Risks and Likely Impacts on West Coast Salmon

Adult coho salmon returns from the fall/winter of 2015–2016 and in the fall/winter of 2016–2017 have likely been negatively impacted by poor stream and ocean conditions. Adult Chinook salmon (and steelhead) returns for the fall/winter 2015–2016 and for the next two to three years (depending on ocean residence times, maturing in 2016, 2017, and 2018) have likely been negatively impacted by poor stream or ocean conditions.

Typical of El Niño winters, there was a more coastally oriented warming of the northeast Pacific in winter 2016 that persisted into early spring 2016. Spring 2016 ocean migrants will likely encounter an ocean strongly influenced by (if not dominated by) a subtropical food-web that favors poor early marine survival for both coho salmon and Chinook salmon.

Summary

Four consecutive years of drought (2012−2015) and the past two years (2014−2015) of exceptionally high air, stream, and upper ocean temperatures have together likely had negative impacts on the freshwater, estuary, and marine phases for many populations of Chinook salmon, coho salmon, and steelhead.

3 Southern Oregon / Northern California Coast Recovery Domain

Thomas H. Williams

3.1 Southern Oregon / Northern California Coast Coho Salmon ESU

The geographic setting of the Southern Oregon/Northern California Coho Salmon Evolutionarily Significant Unit (SONCC-Coho Salmon ESU) includes coastal watersheds from the Elk River (Oregon) in the north to the Mattole River (California) in the south. The ESU is characterized by three large basins and numerous smaller basins across a diverse landscape. The Rogue River and Klamath River extend beyond the Coast Range and include the Cascade Mountains. The Eel River basin also extends well inland, including higher elevation inland streams and those that experience drier, warmer summer temperatures. The numerous smaller to medium-sized coastal basins in the ESU experience relatively wet, cool, and temperate as compared to those of the interior sub-basins of the Rogue River, Klamath River, and Eel River basins, which exhibit a range of conditions including snowmelt-driven hydrographs, hot/dry summers, and cold winters. The lower portions of these large basins are more similar to the smaller coastal basins in terms of environmental conditions than they are to their interior sub-basins.

The Technical Recovery Team (TRT) for the SONCC-Coho Salmon ESU prepared two documents intended to guide recovery planning efforts for the ESA-listed coho salmon. The first of these reports described the historical population structure of the ESU (Williams et al. 2006). In general, the historical population structure of coho salmon in the SONCC-Coho Salmon ESU was characterized by small-to-moderate-sized coastal basins where high quality habitat is in the lower portions of the basin and by three large basins where high quality habitat was located in the lower portions, middle portions of the basins provided little habitat, and the largest amount of habitat was located in the upper portions of the sub-basins. The SONCC TRT categorized populations into one of four distinct types based on its posited historical functional role in the ESU:

> *Functionally independent populations*: populations with a high likelihood of persisting over 100-year time scales and that conform to the definition of independent "viable salmonid populations" offered by McElhany et al. (2000).
>
> *Potentially independent populations*: populations with a high likelihood of persisting over 100-year time scales, but that were too strongly influenced by immigration from other populations to be demographically independent.
>
> *Dependent populations*: populations believed to have had a low likelihood of sustaining themselves over a 100-year time period in isolation and that received sufficient immigration to alter their dynamics and extinction risk.
>
> *Ephemeral populations*: populations that were both small enough and isolated enough that they were only intermittently present.

In addition to categorizing individual populations, the population structure report also placed populations into diversity strata, which are groups of populations that likely exhibit genotypic and phenotypic similarity due to exposure to similar environmental conditions or common evolutionary history (Williams et al. 2006). This effort was a prerequisite for development of viability criteria that consider processes and risks operating at spatial scales larger than those of individual populations.

The second TRT report developed a framework for assessing viability of coho populations in the SONCC-Coho Salmon ESU (Williams et al. 2008). This report established biological viability criteria, from which delisting criteria were developed by a federal recovery planning team (NMFS 2014a). These criteria consist of both population-level viability criteria and ESU-level criteria. Application of these criteria requires time series of adult spawner abundance spanning a minimum of four generations for independent populations.

The population viability criteria represent an extension of an approach developed by Allendorf et al. (1997) and include criteria related to population abundance (effective population size), population decline, catastrophic decline, spawner density, and hatchery influence (Table 3.1). In general, the spawner density low-risk criterion, which seeks to ensure a population's viability in terms its ability to fulfill its historical functional role within the ESU, is the most conservative. The ESU-level criteria are intended to ensure representation of the diversity within an ESU across much of its historical range, to buffer the ESU against potential catastrophic risks, and to provide sufficient connectivity among populations to maintain long-term demographic and genetic processes. These criteria are summarized in Table 3.2.

Since the TRT developed viability criteria for the SONCC-Coho Salmon ESU, a NMFS recovery planning team has completed the federal recovery plan for SONCC-Coho Salmon (NMFS 2014a). This plan includes establishment of population-level and ESU-level recovery criteria for independent populations of SONCC-Coho Salmon. These recovery criteria generally follow the viability criteria developed by the TRT, but may deviate slightly for certain populations based on additional analysis.

Application of recovery and viability criteria requires population-level estimates of adult spawner abundance spanning a minimum of four generations for independent populations (Williams et al. 2008). In reality, for most of the coho salmon populations in this ESU, estimates meeting these criteria are lacking. However, since the mid-2000s, implementation of the Coastal Monitoring Plan[4] (CMP) has greatly expanded, and shorter time series of adult spawner abundance are now available for many populations. In a few other areas composite estimates of several populations, or estimates representing only a portion of a population, constitute the best available data. If data collection has occurred in a consistent manner, these shorter time series, composite estimates, or partial population estimates are presented despite the shortcomings, as they provide the only basis for evaluating current viability. However, the reader is cautioned that short-term trends in abundance or abundance indices can be highly misleading given natural variation in environmental conditions in both the freshwater and marine environments.

---

[4] For information on the California Coastal Monitoring Program:
http://www.calfish.org/ProgramsData/ConservationandManagement/CaliforniaCoastalMonitoring.aspx

Table 3.1. Viability criteria for assessing extinction risk for populations of coho salmon (*O. kisutch*) in the Southern Oregon/Northern California Coast Coho Salmon ESU. For a given population, the highest risk score for any category determines the population's overall extinction risk (Williams et al. 2008).

	Extinction risk		
Criterion	High	Moderate	Low
	- any One of -	- any One of -	- all of -
Effective population size[a]	$N_e \leq 50$	$50 < N_e < 500$	$N_e \geq 500$
- or -	*- or -*	*- or -*	*- or -*
Population size per generation	$N_g \leq 250$	$250 < N_g < 2500$	$N_g \geq 2500$
Population decline	Precipitous decline[b]	Chronic decline or depression[c]	No decline apparent or probable
Catastrophic decline	Order of magnitude decline within one generation	Smaller but significant decline[d]	Not apparent
Spawner density (adults/IP km)	N_a / $IPkm$[e] ≤ 1	$1 < N_a / IPkm <$ MRD[f]	$N_a / IPkm \geq$ MRD[f]
Hatchery Influence			Hatchery fraction <5% *- in addition to above -*
Extinction risk from PVA	≥ 20 % within 20 yrs	≥ 5% within 100 yrs but < 20 % within 20 yrs	< 5 % within 100 yrs[g]

a – The effective population size (N_e) is the number of breeding individuals in an idealized population that would give rise to the same variance in gene frequency under random genetic drift or the same rate of inbreeding as the population under consideration (Wright 1931); total number of spawners per generation (N_g), for SONCC coho salmon the generation time is approximately three years therefore $N_g = 3\ N_a$.
b – Population has declined within the last two generations or is projected to decline within the next two generations (if current trends continue) to annual run size of $N_a \leq 500$ spawners (historically small but stable populations not included) **or** $N_a > 500$ but declining at a rate of ≥10% per year over the last two-to-four generations.
c – Annual spawner abundance (N_a) has declined to ≤ 500 spawners, but now stable **or** number of adult spawners (N_a) > 500 but continued downward trend is evident.
d – Annual spawner abundance decline in one generation < 90% but biologically significant (e.g., loss of year class).
e – *IPkm* = the estimated aggregate intrinsic habitat potential for a population inhabiting a particular watershed (i.e., total accessible km weighted by reach-level estimates of intrinsic potential; see Bjorkstedt et al. [2005] for greater elaboration).
f – Minimum required spawner density (MRD) is dependent on the amount of potential habitat available. Figure 5 of Williams et al. (2008) summarizes the relationship between spawner density and IP km.
g – For population to be considered at low-risk of extinction, all criteria must be satisfied (i.e., not just a PVA). A population viability analysis (PVA) can be also included for consideration, but must estimate an extinction risk <5% within 100 years *and* all other criteria must be met. If discrepancies exist between PVA results and other criteria, results need to be thoroughly examined and potential limitations of either approach should be carefully identified and examined.

Table 3.2. Summary of ESU viability criteria for the Southern Oregon/Northern California Coast Coho Salmon ESU.

ESU viability characteristic	Criteria
Representation	1. All diversity strata should be represented by viable populations.
Redundancy and connectivity	2.a. At least 50% of historically independent populations in each diversity stratum should be demonstrated to be at low risk of extinction according to the population viability criteria. For strata with three or fewer independent populations, at least two populations must be viable.
	AND
	2.b. Total aggregate abundance of the populations selected to satisfy 2a must meet or exceed 50% of the aggregate viable population abundance predicted for the stratum based on the spawner density.
	3. All dependent and independent populations not expected to meet low-risk threshold within a stratum should exhibit occupancy indicating sufficient immigration is occurring from the "core populations".
	4. The distribution of extant populations, both dependent and independent, needs to maintain connectivity across the stratum as well as with adjacent strata.

ESU Boundary Delineation

The SONCC-Coho Salmon ESU currently includes populations spawning from the Elk River (Oregon) in the north to the Mattole River (California) in the south, inclusive. New genetic data are available from collections in 2003 from populations in California (Gilbert-Horvath et al. in press). These recent genetic data do not suggest the need for a re-examination of the boundaries between the Central California Coast Coho Salmon ESU and the SONCC-Coho Salmon ESU. These data show clear separation between populations north and south of Punta Gorda, the current southern boundary of the ESU. The Biological Review Team for the Oregon Coast Coho Salmon ESU reviewed genetic data and concluded that a reconsideration of the ESU boundary the between the SONCC-Coho Salmon ESU and Oregon Coast Coho Salmon ESU was not necessary (Stout et al. 2010). In 2015, a new sampling effort was conducted to resample all sites sampled in 2003 California-wide survey (Gilbert-Horvath et al. in press) and included samples from populations located in the Oregon portion of the SONCC-Coho Salmon ESU. These corresponding analyses are currently underway and therefore are not available for consideration at this time.

Summary of Previous Assessments

Status reviews by Weitkamp et al. (1995) and Good et al. (2005) concluded that the SONCC-Coho Salmon ESU was likely to become endangered. Risk factors identified in these early status reviews included severe declines from historical run sizes, the apparent frequency of local extinctions, long-term trends that were clearly downward, and degraded freshwater habitat and associated reduction in carrying capacity.

In the most recent viability assessment, Williams et al. (2011) reported that although long-term data on coho salmon abundances in the SONCC-Coho Salmon ESU were scarce, all available evidence from shorter-term research and monitoring efforts indicated that conditions had worsened for populations in this ESU since review by Good et al. (2005). Williams et al. (2011) concluded that the SONCC-Coho Salmon ESU was likely to become endangered. The apparent negative trends across the ESU were of great concern as was the lack of information to determine if there had been improvement in freshwater habitat and survival. However, the negative trends were considered in the context of the apparent low marine survival during the period that likely contributed to the observed declines.

New Data and Updated Analyses

Abundance and Trends

Quantitative population-level estimates of adult spawner abundance spanning more than 9–12 years are scarce for independent or dependent populations of coho salmon in the SONCC ESU, although monitoring in California has improved considerably since the 2010 viability assessment as a result of the implementation of the CMP across the California portion of the ESU. The CMP framework provides population abundance estimates at the appropriate spatial scale (i.e., population unit) based on redd counts from surveys of stream reaches selected according to a Generalized Randomized Tessellation Survey (GRTS) design. Redd counts are then expanded to adult estimates based on spawner:redd ratios determined at a network of life-cycle monitoring stations (LCMs). Although only estimates of redds are presented in this assessment of the SONCC-Coho Salmon ESU, these estimates still provide a better basis for assessing viability compared with previous reviews and will increase greatly in value as these time series become longer and we gain a better understanding of the relationship between spawner:redd ratios among populations and among years within a population. Although only one of the time series of abundance meet the requisite four generations called for by the TRT for application of viability criteria, all still provide a substantially better basis for assessing viability compared with previous reviews and will increase greatly in value as these time series become longer. In addition, ongoing weir-based estimates are available for population units in the Klamath Basin (Scott and Shasta rivers), our longest time series sets for this ESU.

Unfortunately, the few estimates available at the population unit spatial scale from the Oregon portion of the ESU for the 2010 assessment are no longer collected and therefore no estimates at the population spatial scale are available for Oregon populations (Sounhein et al. 2014). The estimate of Rogue River coho salmon, that is a composite of several population units, continues to be collected and is extremely valuable.

In California, there are seven independent populations currently monitored at the "population unit" scale, although only the video weir count from the Shasta River is of the duration to satisfy viability criteria (12 years) and is a direct count of fish passing the weir, and not an estimate of adult escapement into the Shasta River (Table 3. 3, Figure 3.1). Of great concern is the extremely low numbers of fish "estimated" to have passed the weir in 2014 (46 coho salmon) and that only four of those fish were considered to be 3-year olds (Chesney and Knechtle 2015). The adult counts from the Scott (Knechtle and Chesney 2014) and Shasta (Chesney and Knechtle 2015) rivers emphasize the current precarious situation in the Klamath. In particular, the Shasta River count is now 14 years in duration (4+ generations) and from this time series a slight decline is apparent, although the slope of the decline is not significantly different from zero (Figure 3.2). In addition, the number of adult coho salmon counted at the Shasta River weir is less than the depensation threshold of 531 adults (Williams et al. 2008).

The Shasta River and Scott River adult counts represent the longest term population-unit spatial scale monitoring currently underway in the SONCC-Coho Salmon ESU, although with implementation of the CMP, five population units are now being monitored and are providing appropriate data to assess population viability (Table 3.3). There are now four years of data (estimated number of redds) for the Smith River, Redwood Creek, Humboldt Bay, and South Fork Eel River population units, although only the first two years of data were available for the Smith River at the time of this assessment. The Mattole River population has a time series of two years and has the lowest estimated number of redds (47) of any of the five new time series available.

Trends in abundance were only calculated for those populations where at least six years of data were available (Table 3.3). The slope of the trend line for both the Shasta River and Scott River did not differ from zero. For the next assessment in 2020, the Scott River will have more than 12 years of data. In addition, the time series information for the Smith River, Redwood Creek, Humboldt Bay, South Fork Eel River, and Mattole River will all be at least two generations in length (six years) and there will be more information on trends in abundance available for other California populations.

Besides the population-unit spatial scale estimate that are required to appropriately assess population viability, there are two other data sets that provide insight into the condition of coho salmon in the ESU, although at spatial scales that do not allow for assessing population viability.

An estimate of spawners over the past 13 years in Freshwater Creek, a Humboldt Bay tributary, includes estimates from 2002–2003 to 2013–2014 with a trend that is not significantly different than zero ($p > 0.07$) over the 13-year period (Figures 3.3 and 3.4; Table 3.4). The Freshwater Creek monitoring site is of particular interest because of the presence of a LCM operated as outlined in the CMP (Ricker and Anderson 2014). This LCM provides critical data to understand the relationships between redds counts and estimated adult escapement – a critical relationship to understand as CMP efforts currently focus on redd counts for many practical reasons. In addition, this and other

Table 3.3. Viability metrics for independent populations of coho salmon (*O. kisutch*) in the SONCC-Coho Salmon ESU. NA indicates not available or applicable; dash (-) indicates no estimate of appropriate spatial scale or sampling design for viability analysis. Trends are shown only for populations where time series is at least six years; **bold** indicates significant trend.

Stratum/population	Yrs	$\overline{N}_{a(arith)}$	$\overline{N}_{a(geom)}$	$\overline{N}_{g(harm)}$	$\hat{C}$	$\hat{T}$ (95% CI)
Northern Coastal Basins						
Elk River						
Lower Rogue River						
Chetco River						
Winchuck River						
Central Coastal Basins						
Smith River[a,b] *(redd estimate)*	2	355	331	NA	NA	-
Lower Klamath River						
Redwood Creek[b,c] *(redd estimate)*	4	529	516	NA	NA	-
Maple Creek/Big Lagoon[d]						
Little River						
Mad River						
Southern Coastal Basins						
Humboldt Bay tributaries[b,e] *(redd estimate)*	4	1038	919	NA	NA	-
Low. Eel/Van Duzen rivers						
Bear River						
Mattole River[b,f] *(redd estimate)*	2	47	46	NA	NA	-
Interior – Rogue						
Illinois River						
Mid. Rogue/Applegate rivers						
Upper Rogue River						
Interior – Klamath						
Middle Klamath River						
Upper Klamath River						
Salmon River						
Scott River[g] *(video weir – adults)*	8	810	404	1713	NA	0.145 (-0.389, 0.678)
Shasta River[h] *(video weir – adults)*	14	127	84	261	0.81	-0.094 (-0.231, 0.044)
Interior – Trinity						
South Fork Trinity River						
Lower Trinity River						
Upper Trinity River						

Table 3.3. continued.

Stratum/population	Yrs	$\overline{N}_{a(arith)}$	$\overline{N}_{a(geom)}$	$\overline{N}_{g(harm)}$	$\hat{C}$	$\hat{T}$ (95% CI)
Interior – Eel						
South Fork Eel River[b,i] *(redd estimate)*	4	1347	1310	NA	NA	-
Mainstem Eel River						
North Fork Eel River[d]						
Middle Fork Eel River[d]						
Middle Mainstem Eel River						
Upper Mainstem Eel River[d]						

a – Data from Garwood and Larson (2014). Data available for 2011 and 2012, data for 2013 and 2014 not available at time of analysis.
b – Redd estimate, not adult escapement estimate.
c – Data from Ricker et al. (2014a, 2014b, 2014c, and 2014d); data from 2010 to 2013.
d – Population unit designated by Williams et al. (2006 and 2008), not included in NMFS (2014a).
e – Data from Ricker et al. (2015e, 2015f, 2015g, and 2015h); data from 2010 to 2013.
f – Data from Ricker and Lindke (2014) and Ricker et al. (2014e); data for 2011 and 2012.
g – Data from Knechtle (2015), data from 2007 to 2014.
h – Data from Knechtle (2015), data from 2001 to 2014.
i – Data from Ricker et al. (2015a, 2015b, 2015c, and 2015d); data from 2010 to 2013.

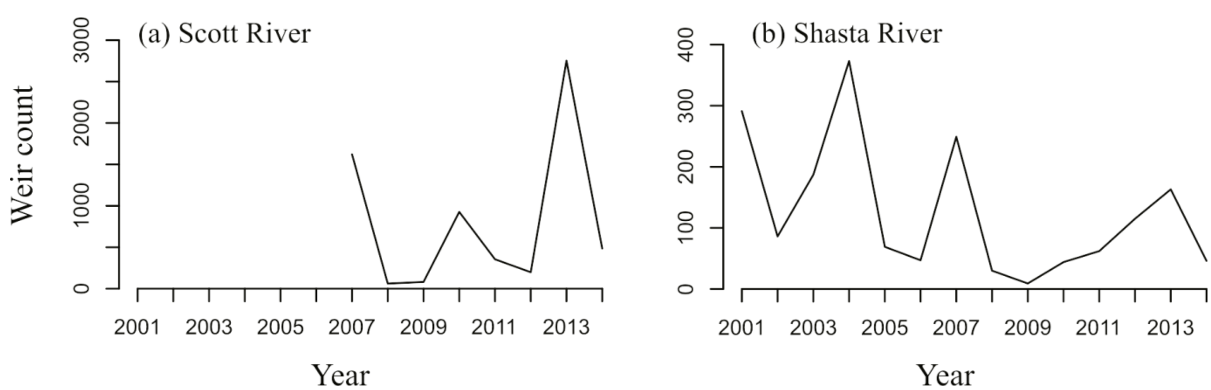

Figure 3.1. Video weir counts of adult coho salmon in a) Scott River for 2007 to 2014 and b) Shasta River for 2001 to 2014 (Knechtle 2015).

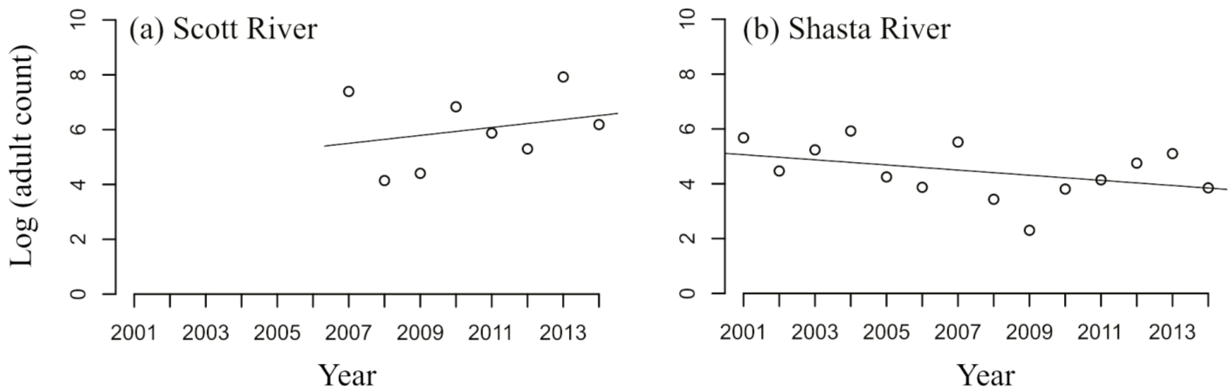

Figure 3.2. Trends (log adult counts at video weir) for independent coho salmon populations in a) Scott River from 2007 to 2014 and b) Shasta River from 2001 to 2014 (Knechtle 2015).

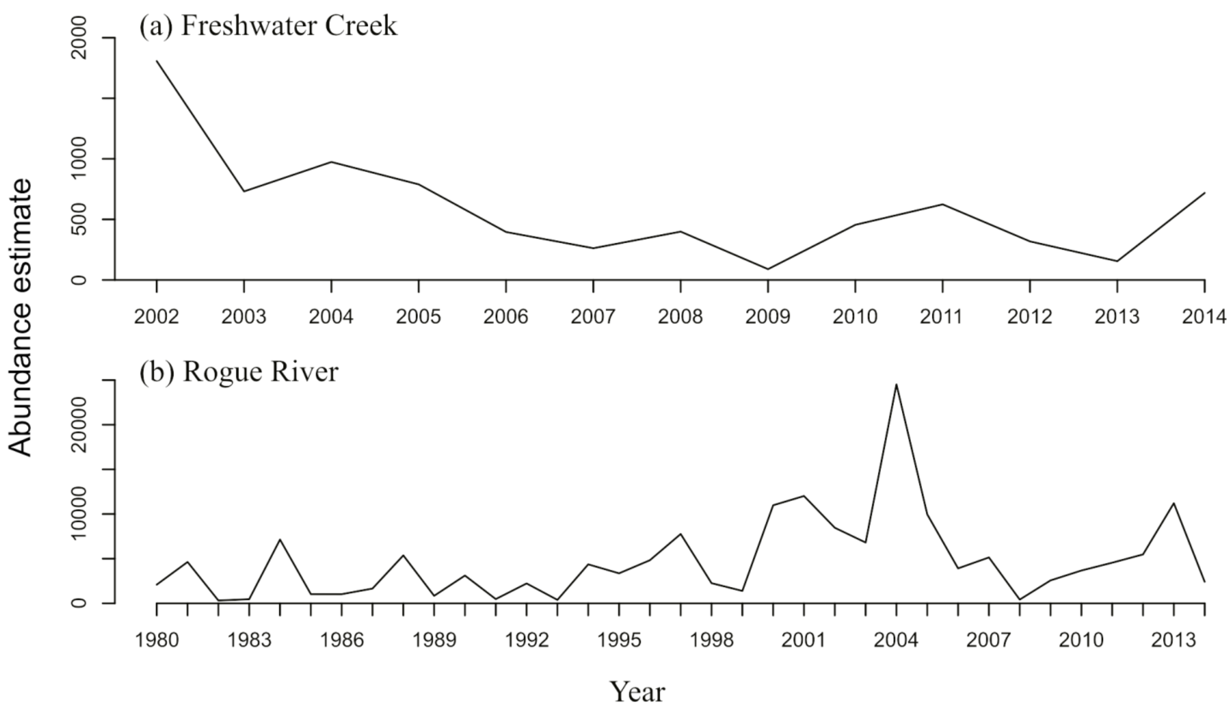

Figure 3.3. Estimated adult coho salmon in a) Freshwater Creek for 2002 to 2014 and b) Rogue River for 1980 to 2014 (Freshwater Creek data from Ricker 2015, Rogue River data from Confer (2015).

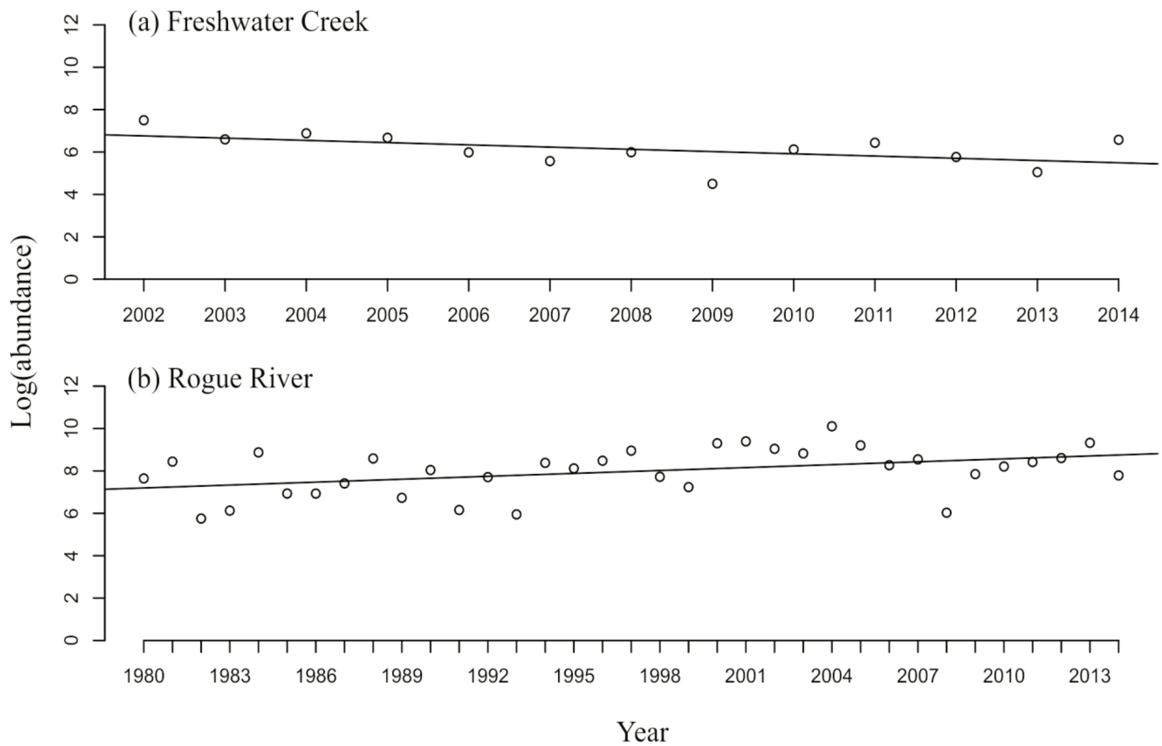

Figure 3.4. Trends (log abundance) of coho salmon in a) Freshwater Creek for 2002 to 2014 and b) Rogue River for 1980 to 2014 (Freshwater Creek data from Ricker 2015, Rogue River data from Confer (2015).

LCMs will provide estimates of marine survival that will provide context when evaluating trends in abundance and effectiveness of restoration activities (Figure 3.5).

The only estimate available to assess the viability of coho salmon in the Oregon portion of the SONCC-Coho Salmon ESU is from the Rogue River. These estimates are derived from mark-recapture estimates based on returns to Cole Rivers Hatchery expanded by the mark rate observed at Huntley Park. (Confer 2015). The Huntly Park seine estimates provide the best overall assessment of coho salmon spawner abundance in the basin (Good et al. 2005). Four independent populations contribute to this count (Lower Rogue River, Illinois River, Middle Rogue and Applegate rivers, and Upper Rogue River), which has had a significant positive trend ($p = 0.01$) over the past 35 years and a non-significant negative trend ($p > 0.05$) over the past 12 years or four generations (Table 3.4; Figures 3.3 and 3.4).

No extensive and systematic survey of presence of coho salmon has been conducted in the SONCC-Coho Salmon ESU in the past 10 years. Garwood (2012) developed a criteria to develop a list of historical and recent occurrence of coho salmon in the California portion of the SONCC-Coho Salmon ESU, although brood years evaluated were almost exclusively from 1979 to 2004 and therefore did not include field observations for the most recent three generations. No comparable survey data are available for the period from 2005 to 2014.

Table 3.4. Short- and long-term trends in SONCC-Coho Salmon ESU population abundance based on partial or composite population estimates and population indices. Trends in **bold** are significantly different from 0 ($\alpha = 0.05$).

Spawning tributary (Population)	Years	Data type	Mean (range)	$\hat{T}$ (95% CI)	Data sources
Rogue Basin[a]	12	Composite, mark-recapture	6717 (414 – 24509)	-0.074 (-0.262, 0.150)	Confer 2015
	35		4764 (314 – 24509)	**0.046** (0.011, 0.115)	
Freshwater Creek[b] (Humboldt Bay)	12	Partial pop., weir-carcass mark-recapture	493 (89 – 974)	-0.070 (-0.200, 0.060)	Ricker 2015
	13		594 (89 – 1807)	-0.105 (-0.222, 0.013)	Ricker 2015

a – These estimates are derived from mark-recapture estimates based on returns to Cole Rivers Hatchery expanded by the mark rate observed at Huntley Park. The Oregon Department of Fish and Wildlife staff advises that these data provide a more precise estimate of coho salmon escapement in the Rogue Basin compared to Huntly expansion method used previously (and in 2010 review). Data from Confer (2015).
b – Maximum live/dead counts do not distinguish between natural and hatchery-origin spawners. Counts may include both, particularly in the early part of the time series.

There are three hatcheries in the SONCC Coho Salmon ESU and all three are included in the ESA-listed ESU. The hatcheries include Cole Rivers Hatchery on the Rogue River, Iron Gate Hatchery on the Klamath River, and Trinity River Hatchery on the Trinity River. One key development since the previous assessment in 2010 is the completion of the Hatchery Genetic Management Plan (HGMP) for the Iron Gate Hatchery that moves the operation of this hatchery from a mitigation hatchery to one now operated to protect and conserve the genetic resources of the Upper Klamath population unit of the SONCC-Coho Salmon ESU. Included in the HGMP are defined monitoring and evaluation activities to evaluate effects of the hatchery activities on the abundance, productivity, spatial structure, and diversity, and the magnitude or relative impact of the hatchery program on other actions that influence the SONCC-Coho Salmon ESU. The implementation of the HGMP is considered a positive step towards meeting viability targets for the Upper Klamath population unit, the diversity stratum, and the ESU. A HGMP is being developed for the Trinity River Hatchery and is not in place at this time. Cole Rivers Hatchery is operated as a harvest program (ODFW 2015) used for augmentation of fishing and harvest opportunities, and mitigation for the loss of habitat resulting from dam construction in the Rogue and Applegate rivers. A HGMP was completed in 1999. The hatchery stock is managed as an integrated stock. Approximately 75,000 smolts are released on-site, all fish are fin-clipped and 25,000 are coded-wire tagged (ODFW 2015). The coho salmon program at Cole Rivers Hatchery does provide monitoring opportunities related to ocean distribution and harvest. Future development of a HGMP for Trinity River Hatchery will help insure that hatchery operations for coho

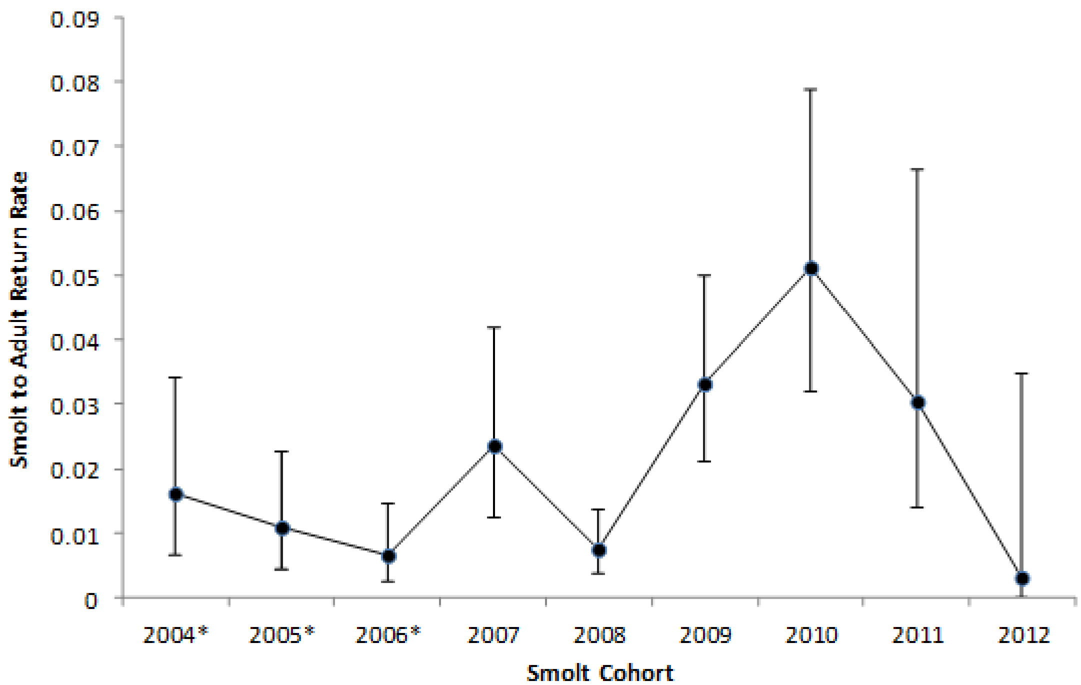

Figure 3.5. Smolt to adult return rates and 95% confidence levels for Freshwater Creek coho salmon smolt cohorts 2004 to 2012 from Ricker and Anderson (2014). Estimates for cohorts 2004-2006 based on smolts trapped at different location than estimates for cohorts 2005-2012 (see Ricker and Anderson 2014).

salmon are focused on aspects that protect and conserve the genetic resources of the local population units of the SONCC-Coho Salmon ESU and include defined monitoring and evaluation activities to evaluate effects of the hatchery activities on the abundance, productivity, spatial structure, and diversity, and the magnitude or relative impact of the hatchery program on other actions that influence coho salmon in this ESU.

Harvest Impacts[5]

Coho salmon from this ESU are primarily distributed off the coast of California and southern Oregon. Because coho salmon-directed fisheries and coho salmon retention have been prohibited off the coast of California since 1996, the ocean exploitation rate of SONCC-Coho Salmon is generally low and attributable to non-retention impacts in California and Oregon Chinook salmon-directed fisheries, impacts in Oregon mark-selective coho salmon fisheries (primarily non-retention), and impacts in Oregon non-mark selective fisheries.

[5] Harvest impacts section prepared by Michael O’Farrell.

Natural-origin Rogue/Klamath basin coho salmon ocean exploitation rates have been estimated for years 1986–2014 using backward runs of the Fishery Regulation Assessment Model (FRAM) (L. LaVoy and R. Kope, NMFS, personal communication). These estimates are the best available measure of the SONCC-Coho Salmon ESU ocean exploitation rate (Figure 3.6). This rate has been low and relatively stable since the early 1990s (average of 5.3% for years 1994–2014), which contrasts sharply with the much higher rates estimated for the 1980s and early 1990s (average of 50.8% between 1986 and 1993).

Freshwater recreational fishery impacts on SONCC-Coho Salmon are likely relatively low given California's statewide prohibition of coho salmon retention, and normally only mark-selective coho salmon retention in the Oregon portion of this ESU. Klamath basin tribes (Yurok, Hoopa, and Karuk) currently harvest a relatively small number of coho salmon for subsistence and ceremonial purposes (CDFG 2002). The Yurok fishery estimated harvest rate averaged 3.4% for the 1994–2014 period, and 2.3% for the 2011–2014 period (Williams 2015). The harvest rates reported in Williams (2015) are likely biased high because little escapement and harvest monitoring occur in the Klamath Basin, precluding a complete estimate of run size. Harvest rate estimates for the other two tribal fisheries are not available.

In summary, the available information indicates that the level of SONCC-Coho Salmon ESU fishery impacts has not changed appreciably since the 2010 salmon and steelhead assessment (Williams et al. 2011).

Summary and Conclusions

Although long-term data on coho salmon abundance in the SONCC-Coho Salmon ESU are scarce, all available evidence from available trends since 2011 assessment (Williams et al. 2011) indicate little change since the 2011 assessment. The two population-unit scale time series for the ESU both have a trend slope not different than zero. The composite estimate for the Rogue Basin populations was not significantly different from zero ($p > 0.05$) over the past 12 years and significantly positive over the 35 years of the data set ($p = 0.01$). The continued lack of appropriate data remains a concern, although the implementation of the CMP for California populations is an extremely positive step in the correct direction in terms of providing the types of information required to adequately assess and evaluate population and ESU viability. The lack of population spatial scale monitoring sites in Oregon is of great concern and increases the uncertainty when assessing viability. Additionally, it is evident that many independent populations are well below low-risk abundance targets, and several are likely below the high-risk depensation thresholds specified by the TRT and the Recovery Plan (NMFS 2014a). Though population-level estimates of abundance for most independent populations are lacking, it does not appear that any of the seven diversity strata currently supports a single viable population as defined by the TRT's viability criteria, although all diversity strata are occupied.

In addition to the implementation of population monitoring in California through the CMP, the implementation of Life-Cycle Monitoring stations is also an extremely positive development and will greatly contribute to estimating freshwater and marine survival,

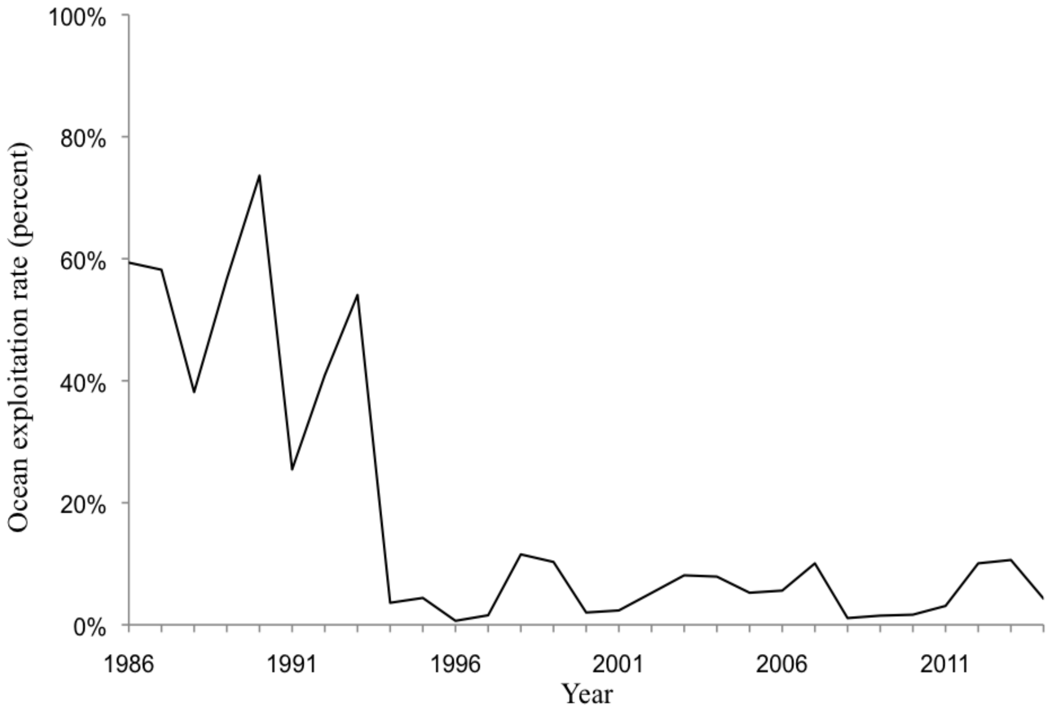

Figure 3.6. Natural-origin Rogue/Klamath coho salmon ocean exploitation rate estimates for years 1986 – 2014 (L. LaVoy and R. Kope, NMFS, personal communication). calibrating various sampling methods, and providing platforms for needed research to further develop appropriate conservation and recovery efforts.

The SONCC-Coho Salmon ESU is currently considered likely to become endangered. Of particular concern is the low number of adults counted entering the Shasta River in 2014–2015. The lack of increasing abundance trends across the ESU for the populations with adequate data are of concern. Moreover, the loss of population spatial scale estimates from coastal Oregon populations is of great concern. The new information available since Williams et al. (2011) while cause for concern, does not appear to suggest a change in extinction risk at this time.

4 North-Central California Coast Recovery Domain

Brian C. Spence

The North-Central California Coast Recovery Domain encompasses the geographic region from Redwood Creek (Humboldt County) south to Aptos Creek (Santa Cruz County) inclusive. Two salmon Evolutionarily Significant Units (ESUs) and two steelhead Distinct Population Segments (DPSs) lie wholly within this region: California Coastal Chinook Salmon, Central California Coast Coho Salmon, Northern California Steelhead, and Central California Coast Steelhead.

The Technical Recovery Team (TRT) for the North-Central California Coast Recovery Domain prepared two documents intended to guide recovery planning efforts for the ESA-listed salmonids within the domain. The first of these reports described the historical population structure of the four listed ESU/DPSs within the recovery domain (Bjorkstedt et al. 2005). Within this document, the TRT categorized each population into one of three distinct types based on its posited historical functional role:

> *Functionally independent populations*: populations with a high likelihood of persisting over 100-year time scales and that conform to the definition of independent "viable salmonid populations" offered by McElhany et al. (2000).
>
> *Potentially independent populations*: populations with a high likelihood of persisting over 100-year time scales, but that were too strongly influenced by immigration from other populations to exhibit independent dynamics.
>
> *Dependent populations*: populations that had a substantial likelihood of going extinct within 100-year time period in isolation, yet received sufficient immigration to alter their dynamics and reduce their risk of extinction.

In addition to categorizing individual populations, the population structure report also places populations into *diversity strata*, which are groups of populations that likely exhibit genotypic and phenotypic similarity due to exposure to similar environmental conditions or common evolutionary history (Bjorkstedt et al. 2005; revised in Spence et al. 2008). Here, the TRT set the stage for development of viability criteria that consider processes and risks operating at spatial scales larger than those of individual populations.

The second TRT report proposes a framework for assessing viability of populations and ESU/DPSs within the recovery domain (Spence et al. 2008). This report establishes both population-level and ESU/DPS-level biological viability criteria. The population viability criteria developed by the TRTs represent an extension of an approach developed by Allendorf et al. (1997) and include criteria related to population abundance (effective population size), population decline, catastrophic decline, spawner density, and hatchery influence (Table 4.1). In general, the spawner density low-risk criterion, which seeks to ensure a population's ability to fulfill its historical functional role within the ESU, is the most conservative, and preliminary viability targets for each population were determined primarily by this criterion. The ESU-level criteria are intended to ensure representation of the diversity within an ESU/DPS across much of its historical range, to buffer the

Table 4.1. Criteria for assessing the level of risk of extinction for populations of Pacific salmonids. Overall risk is determined by the highest risk score for any category. N_g = generational sum of abundance; N_e = effective population size; and N_a = annual spawner abundance. From Spence et al. (2008).

Population Characteristic	Extinction Risk: High	Moderate	Low
Extinction risk from population viability analysis (PVA)	≥ 20% within 20 yrs - or any ONE of the following -	≥ 5% within 100 yrs but < 20% within 20 yrs - or any ONE of the following -	< 5% within 100 yrs - or ALL of the following -
Effective population size per generation *-or-* Total population size per generation	$N_e \leq 50$ *-or-* $N_g \leq 250$	$50 < N_e < 500$ *-or-* $250 < N_g < 2500$	$N_e \geq 500$ *-or-* $N_g \geq 2500$
Population decline	Precipitous decline[a]	Chronic decline or depression[b]	No decline apparent or probable
Catastrophic decline	Order of magnitude decline within one generation	Smaller but significant decline[c]	Not apparent
Spawner density	$N_a/IPkm^{d} \leq 1$	$1 < N_a/IPkm <$ MRD[e]	$N_a/IPkm \geq$ MRD[e]
Hatchery influence[f]	Evidence of adverse genetic, demographic, or ecological effects of hatcheries on wild population		No evidence of adverse genetic, demographic, or ecological effects of hatchery fish on wild population

a – Population has declined within the last two generations or is projected to decline within the next two generations (if current trends continue) to annual run size $N_a \leq 500$ spawners (historically small but stable populations not included) *or* $N_a > 500$ but declining at a rate of ≥10% per year over the last two-to-four generations.
b – Annual run size N_a has declined to ≤ 500 spawners, but is now stable *or* run size $N_a > 500$ but continued downward trend is evident.
c – Annual run size decline in one generation < 90% but biologically significant (e.g., loss of year class).
d – *IPkm* = the estimated aggregate intrinsic habitat potential for a population inhabiting a particular watershed (i.e., total accessible km weighted by reach-level estimates of intrinsic potential; see Bjorkstedt et al. [2005] for greater elaboration).

ESU/DPS against potential catastrophic risks, and to provide sufficient connectivity among populations to maintain long-term demographic and genetic processes. These criteria are summarized in Table 4.2.

Since the TRT developed viability criteria for the NCCC Recovery Domain, NMFS recovery planning teams have completed the federal recovery plan for CCC-Coho Salmon (NMFS 2012a). This plan includes establishment of population-level and ESU-level recovery criteria for independent populations of the CCC-Coho Salmon ESU. These

Table 4.2. ESU-level criteria for assessing the level of risk of extinction for Pacific salmonid ESUs. From Spence et al. (2008).

Criterion	Description
Representation	All identified diversity strata that include historical functionally or potentially independent populations within an ESU/DPS should be represented by viable populations for the ESU/DPS to be considered viable -AND- Within each diversity stratum, all extant phenotypic diversity (i.e., major life-history types) should be represented by viable populations
Redundancy and Connectivity	At least 50% of historically independent populations in each diversity stratum must be demonstrated to be at low risk of extinction according to the population viability criteria outlined in Table 1 of Spence et al. (2008) -AND- Within each diversity stratum, the total aggregate abundance of independent populations selected to satisfy this criterion must meet or exceed 50% of the aggregate viable population abundance (i.e., meeting density-based criteria for low risk) for all independent populations Remaining populations, including historical dependent populations and any historical independent populations that are not expected to attain a viable status must exhibit occupancy patterns consistent with those expected under sufficient immigration subsidy arising from the "core" independent populations selected to satisfy the preceding criterion The distribution of extant populations, regardless of historical status, must maintain connectivity within the diversity stratum, as well as connectivity to neighboring diversity strata

recovery criteria generally follow the viability criteria developed by the TRT, but may deviate slightly for certain populations based on additional analysis. Additionally, the plan develops numeric criteria for selected dependent populations. For the purpose of this viability assessment, we use the recovery criteria for CCC-Coho Salmon outlined in the recovery plan as the benchmark for assessing viability.

A draft multispecies recovery plan covering the CC-Chinook Salmon ESU , NC-Steelhead DPS, and CCC-Steelhead DPS is currently undergoing public review. Because the recovery criteria specified in this draft plan are subject to change, we have used the TRT's viability criteria as the basis for evaluating viability in this review.

Application of recovery and viability criteria requires population-level estimates of adult spawner abundance spanning a minimum of four generations for independent populations (Spence et al. 2008). In reality, for most of the salmon and steelhead populations in this recovery domain, estimates meeting these criteria are lacking. However, since the mid-2000s, implementation of the Coastal Monitoring Plan (CMP) has greatly expanded, and shorter time series of adult spawner abundance are now available for many watersheds. In other areas, indices of spawner abundance or local population estimates representing only a portion of the population constitute the best available data. If data collection has occurred in a consistent manner, these shorter time series, indices, or partial population estimates are presented herein despite the shortcomings, as they provide the only basis for evaluating current viability. However, the reader is cautioned that short-term trends in abundance or abundance indices can be highly misleading given natural variation in environmental conditions in both the freshwater and marine environments. A complete list of data sources for the analysis of ESU/DPSs in the North-Central California Coast Recovery Domain can be found in Appendix A.

4.1 Central California Coast Coho Salmon ESU

ESU Boundary Delineation

The initial status review for the Central California Coast (CCC) Coho Salmon ESU (Weitkamp et al. 1995) defined the ESU as populations from Punta Gorda southward to and including the San Lorenzo River. Since that time, the boundary has been extended southward to include Soquel and Aptos creeks (77 FR 19552) based on analysis of historical and recent evidence of occurrence as well as environmental conditions in these two watersheds (Spence et al. 2011). Successful reproduction of coho salmon in Soquel Creek was again documented in summer of 2015 (B. Spence and J. Kiernan, NMFS SWFSC, personal communication), which supports the boundary extension.

In 2003, NMFS Southwest Fisheries Science Center conducted an extensive genetic survey of coho salmon populations in coastal California. Genetic samples were taken from juvenile coho salmon collected at 30 sites in 23 different watersheds spanning the SONCC and CCC ESUs. Multiple analyses of microsatellite data provided consistent and strong support for the current ESU boundary at Punta Gorda (Gilbert-Horvath et al. in press).

Summary of Previous Assessments

Status reviews by Weitkamp et al. (1995) and Good et al. (2005) both concluded that the CCC-Coho Salmon ESU was in danger of extinction. These reviews cited concerns over low abundance and long-term downward trends in abundance throughout the ESU, as well as extirpation or near extirpation of populations across most of the southern two-thirds of the ESU’s historical range, including several major river basins. They further cited as risk factors the potential loss of genetic diversity associated with range reductions or loss of one or more brood lineages, coupled with historical influence of hatchery fish (Good et al. 2005). NMFS initially listed CCC-Coho Salmon ESU as threatened in 1996 (61 FR 56138), but changed the status to endangered in 2005 (70 FR 37160). In the most recent assessment, Spence and Williams (2011) concluded that conditions of populations in the CCC-Coho Salmon ESU had worsened since 2005, noting negative trends for most independent and dependent populations for which longer term monitoring data were available, and the near complete collapse and loss of genetic diversity for populations in the Santa Cruz Mountains Diversity Stratum. NMFS subsequently concluded that the CCC-Coho Salmon ESU remained endangered (77 FR 19552).

New Data and Updated Analyses

Abundance and Trends

Information on population status and trends for CCC-Coho Salmon has improved considerably since the 2010 viability assessment due to recent implementation of the Coastal Monitoring Plan across significant portions of the ESU. Population estimates are based on redd counts from surveys of stream reaches selected according to a Generalized Randomized Tessellation Survey (GRTS) design. Redd counts are then expanded to adult estimates based on spawner:redd ratios determined at a network of life-cycle monitoring stations. Although many of the time series of abundance do not currently meet the requisite four generations called for by the TRT for application of viability criteria, they still provide a substantially better basis for assessing viability compared with previous reviews and will increase greatly in value as these time series become longer. Below, we review available information for each of the four diversity strata for which recovery criteria have been proposed.

Lost Coast – Navarro Point Stratum. Population-level estimates of adult abundance are now available for all four independent populations and as well as seven dependent populations of coho salmon within this stratum. For the Noyo River, Pudding Creek, Caspar Creek, and Little River, these time series span from 12–15 years, whereas for the remainder of populations, the time series are shorter (3–6 years). Recent population estimates indicate that population sizes are currently from 4% (Big River) to 13% (Noyo River) of the proposed recovery targets (Table 4.3). One population (Big River) is below the high-risk depensation threshold ($D_{dep} = 0.6$) and a second (Albion River) is right at the threshold ($D_{dep} = 1.0$). Recent trends are variable, with the Ten Mile River, Big River, and Albion River showing positive but non-significant trends ($p > 0.10$) and the Noyo River showing essentially no trend (Table 4.3; Figures 4.1a-d; Figures 4.2a-d). Importantly, the Noyo River time series is six years longer than the other populations, and the trend for the past 5–6 years has been positive.

For dependent populations, Pudding Creek and Caspar Creek appear to be the strongest populations, with average returns of 417 and 115 adults, respectively over the last 14–15 years (Table 4.4). These numbers are approximately 42% and 26% of recovery targets, respectively. However, trends for these two populations, as well as for the Little River population, for the period of record are negative and significant ($p < 0.05$) (Table 4.4; Figure 4.3b-d; Figure 4.4b-d). Very low numbers of coho salmon have been observed in Usal Creek and Big Salmon Creek, and no coho salmon have been observed in four years of record for Wages Creek and Cottaneva Creek (Table 4.4).

Table 4.3. Viability metrics for independent populations of coho salmon in the CCC-Coho Salmon ESU. NA indicates not available or applicable. Trends are shown only for populations where time series is at least six years; bold indicates significant trend. IPkm includes only habitats that are currently accessible. $N_{a(arith)}$ target refers to target identified in recovery plan (NMFS 2012a).

Stratum/population	Years	$\overline{N}_{a(arith)}$	$\overline{N}_{a(geom)}$	$\overline{N}_{g(harm)}$	$\hat{C}$	$\hat{T}$ *(95% CI)*	IPkm	$\hat{D}_{dep}$	$\hat{D}_{ssd}$	$\hat{D}_{ssd}$ target	$\overline{N}_{a(arith)}$ target
Lost Coast - Navarro Pt											
Ten Mile River[a]	6	359	69	1163	NA	0.300 (-1.794, 2.393)	105.1	1.9	3.4	34.9	3700
Noyo River[a]	12	539	455	1182	0.50	-0.020 (-0.114, 0.073)	118.0	2.4	4.4	34.0	4000
Big River[a]	6	220	183	609	NA	0.224 (-0.134, 0.582)	191.8	0.6	1.1	28.9	5500
Albion River[a]	6	188	21	328	NA	0.243 (-1.798, 2.285)	59.2	1.0	3.2	38.1	2300
Navarro Pt - Gualala Pt											
Navarro River[a]	6	257	102	867	NA	-0.645 (-2.158, 0.868)	201.0	1.0	1.3	28.3	5700
Garcia River[a]	6	64	18	166	NA	-0.276 (-1.766, 1.214)	76.0	0.4	0.8	36.9	3700
Gualala River	-	-	-	-	-	-	251.6	-	-	24.8	6200
Coastal											
Russian River[b]	5	364	-	-	-	-	757.4	-	-	20.0	10100
Walker Creek	-	-	-	-	-	-	76.2	-	-	36.9	2600
Lagunitas Creek[c]	17	512	408	1109	0.85	-0.063 (-0.140, 0.014)	70.4	1.8	6.9	37.3	2600
Santa Cruz Mtn											
Pescadero Creek[d]	4	0	1	0	NA	NA	60.6	0	0	38.0	2300
San Lorenzo River[d]	3	1	1	3	NA	NA	126.4	0	0	33.4	3800

a – Numbers indicate the estimated number of adults based on fish/redd expansions from life-cycle monitoring stations.
b – Numbers indicate expanded estimates derived from multiple methods (spawner surveys, adult traps, video counts, PIT tag detections, hatchery returns, and independent observations, as well as inference from juvenile observations and downstream migrant trapping). As methods and spatial extent have varied over time, only arithmetic mean is presented as a minimum estimate. give a .
c – Numbers indicate 2x total redd counts. Methods have not yet been developed to derive fish/redd estimates for expansion.
d – Numbers indicate numbers of observed fish (live adults + carcasses). Methods have not yet been developed to derive fish/redd estimates for expansion.

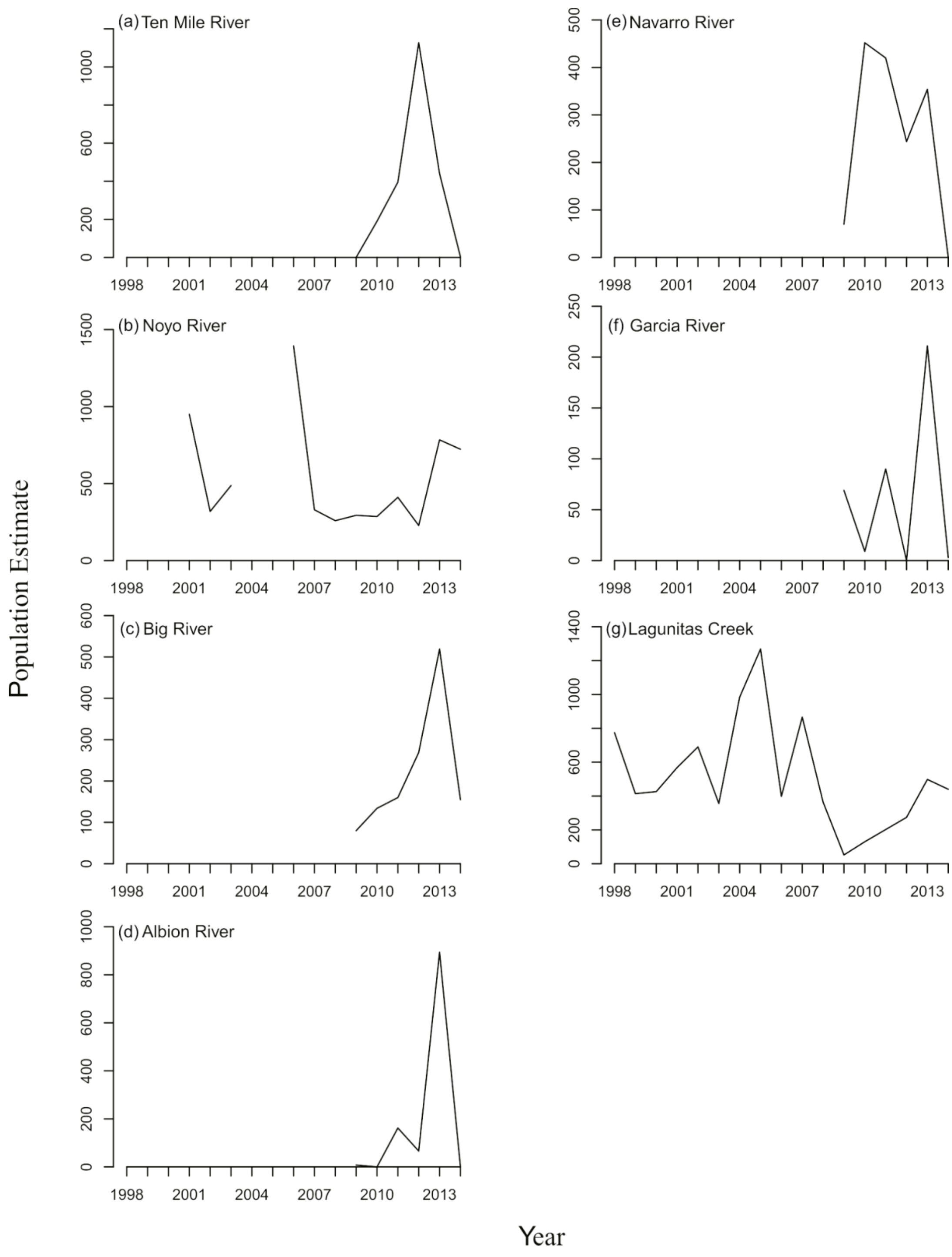

Figure 4.1. Time series of population abundance estimates for independent populations of CCC-Coho Salmon. Values for Lagunitas Creek are two times the total redd count for the watershed. All other estimates are based on fish/redd expansions from life-cycle monitoring stations.

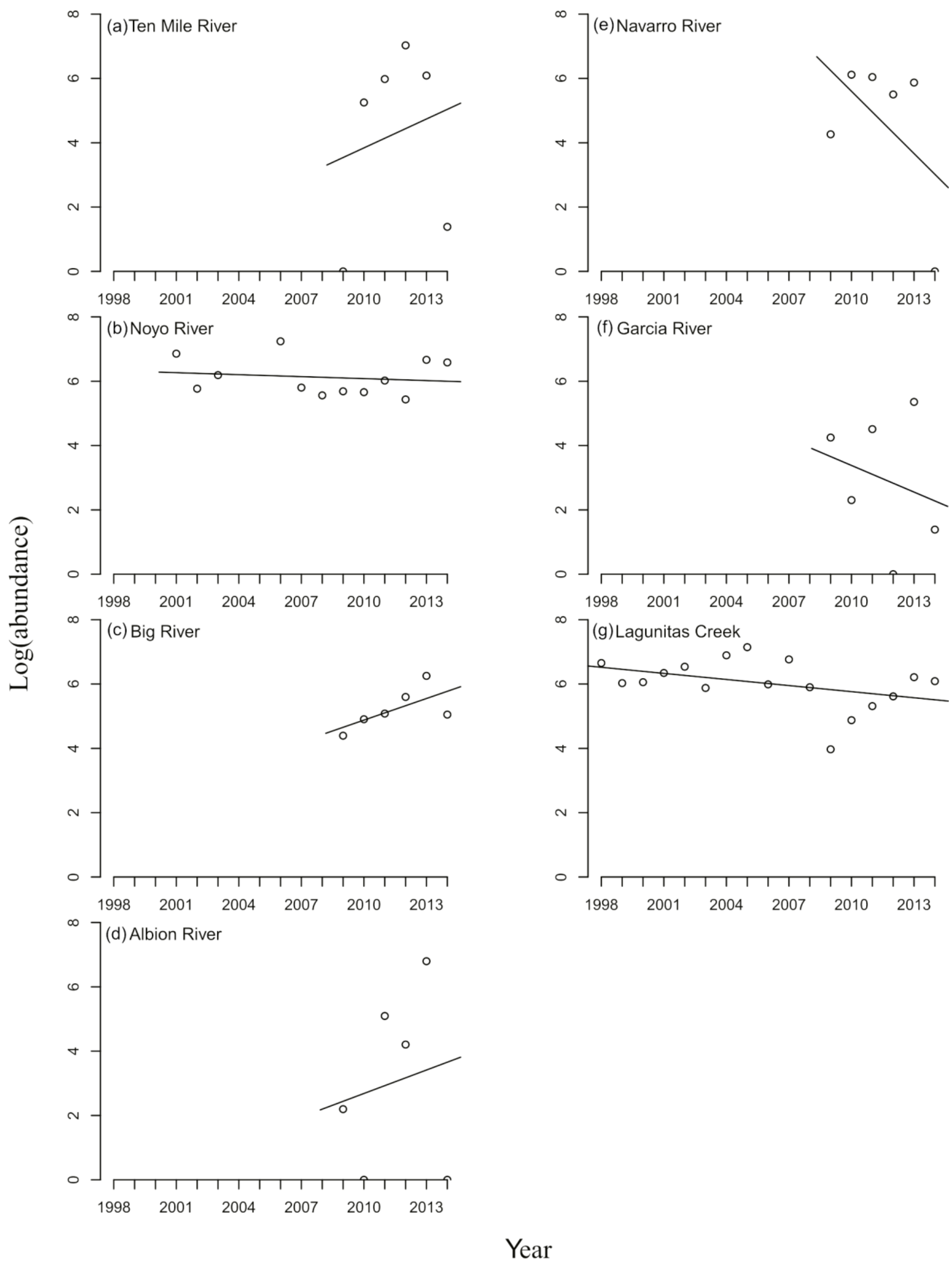

Figure 4.2. Population trends (log abundance) for independent populations of CCC-Coho Salmon. Values for Lagunitas Creek are based on two times the total redd count for the watershed. All other estimates are based on fish/redd expansions from life-cycle monitoring stations.

Table 4.4. Viability metrics for dependent populations of coho salmon in the CCC-Coho Salmon ESU. NA indicates not available or applicable. Trends are shown only for populations where time series is at least six years, **bold** indicates significant trend. $N_{a(arith)}$ target refers to target identified in CCC-coho salmon recovery plan (NMFS 2012a).

Stratum/population	Years	$\overline{N}_{a(arith)}$	$\overline{N}_{a(geom)}$	$\overline{N}_{g(harm)}$	$\hat{T}$ *(95% CI)*	$\overline{N}_{a(arith)}$ target
Lost Coast - Navarro Pt						
Usal Creek	6	6	4	16	-0.142 (-1.031, 0.747)	360
Cottaneva Creek	4	0	NA	NA	NA	469
Wages Creek	4	0	NA	NA	NA	340
Pudding Creek	14	417	184	741	**-0.272 (-0.510, -0.034)**	983
Caspar Creek	15	115	40	86	**-0.304 (-0.447, -0.161)**	435
Little River	15	30	10	19	**-0.236 (-0.361, -0.110)**	NA
Big Salmon Creek	3	6	3	NA	NA	578
Navarro Pt – Gualala Pt						
Greenwood Creek	2	4	3	NA	NA	NA
Elk Creek	2	0	NA	NA	NA	NA
Brush Creek	6	0	NA	NA	NA	NA
Coastal						
Salmon Creek	-	-	-	-	-	1367
Pine Gulch	14	1	2	0	-0.064 (-0.171, 0.043)	394
Redwood Creek	17	47	23	90	-0.105 (-0.229, 0.020)	272
Santa Cruz Mtn						
San Gregorio Creek	-	-	-	-	-	1363
Gazos Creek	3	0	NA	NA	NA	279
Waddell Creek	4	1*	1*	0*	NA	313
Scott Creek	13	71	18	31	-0.095 (-0.380, 0.189)	510
San Vicente Creek	3	2*	2*	6*	NA	105
Soquel Creek	-	-	-	-	-	1122
Aptos Creek	1	0	NA	NA	NA	932

* Low abundances of coho salmon have precluded development of relationships between redd counts and estimated numbers of spawners. Mean values presented reflect numbers of observed fish (live adults plus recovered carcasses).

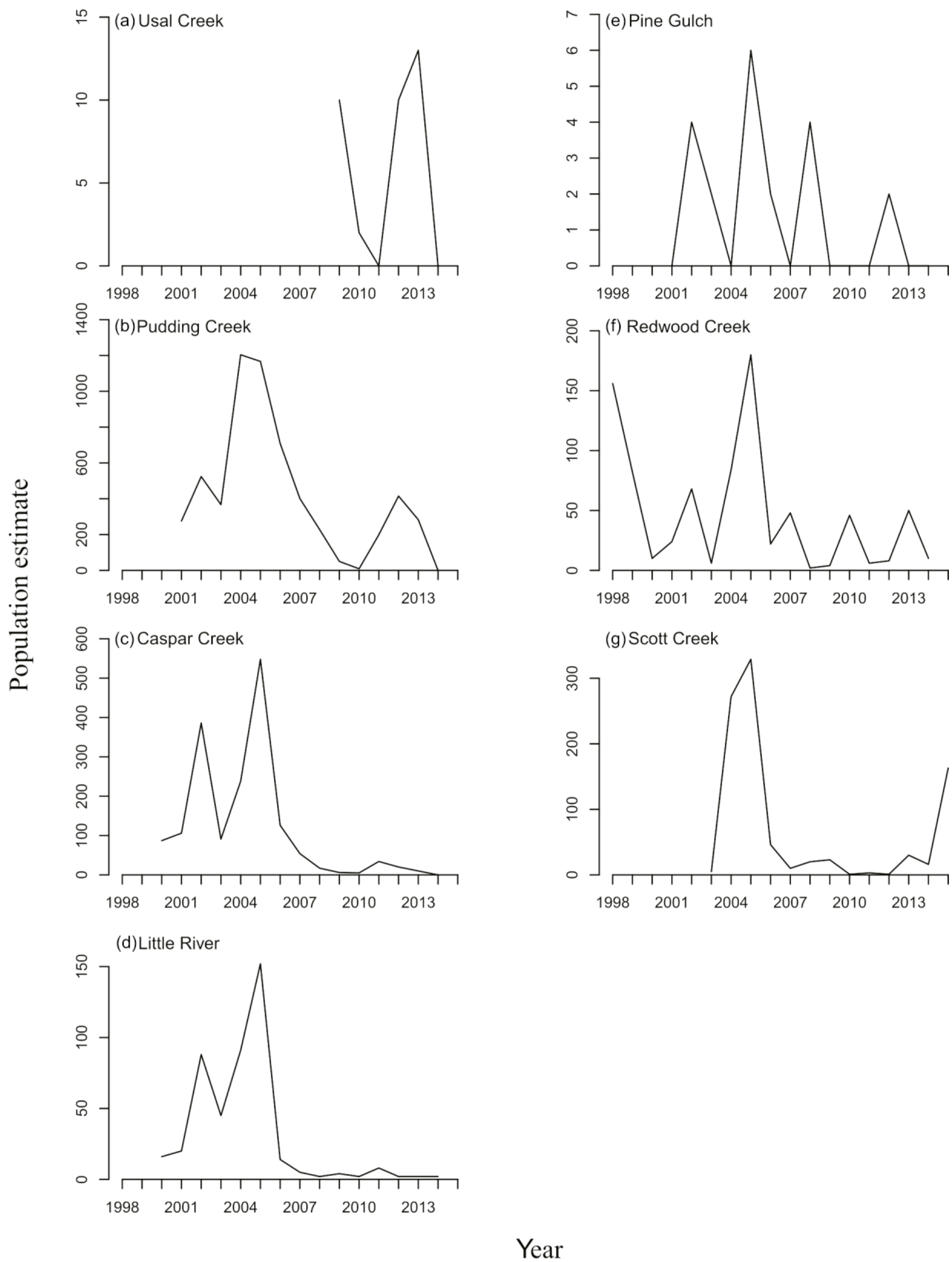

Figure 4.3. Time series of population abundance estimates for dependent populations of CCC-Coho Salmon. Values for Redwood Creek and Pine Gulch are two times the total redd count for the watershed. All other estimates are based on fish/redd expansions from life-cycle monitoring stations.

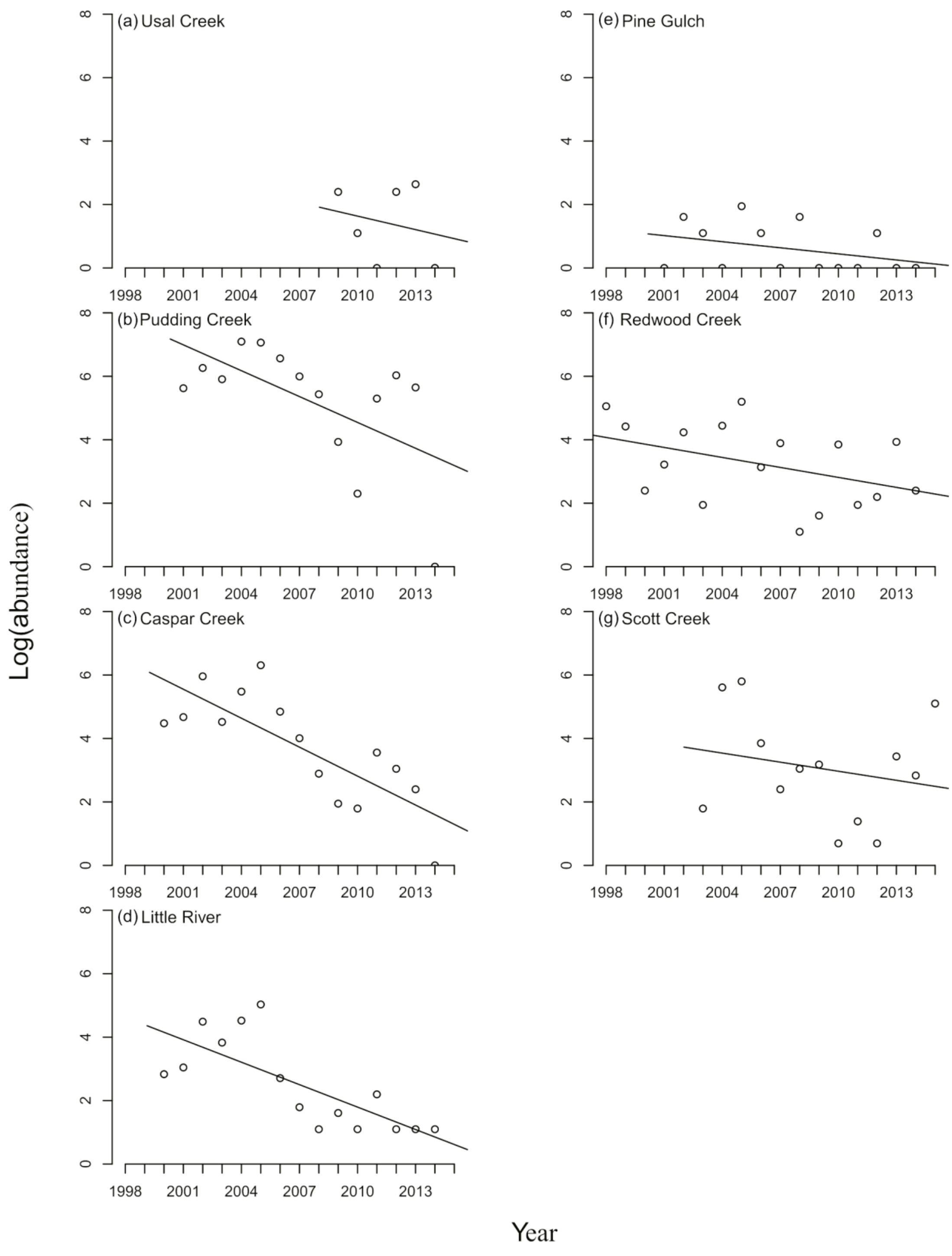

Figure 4.4. Population trends (log abundance) for dependent populations of CCC-Coho Salmon. Values for Redwood Creek and Pine Gulch are based on two times the total redd count for the watershed. All other estimates are based on fish/redd expansions from life-cycle monitoring stations.

Navarro Point – Gualala Point Stratum. Two of three independent populations in this stratum now have time series of adult abundance spanning six years: Navarro River and Garcia River. These data sets indicate that adult population sizes have averaged 257 and 64 fish, respectively (Table 4.3). Both populations are at less than 5% of the recovery targets and are at or below the depensation high-risk threshold. The six-year trend for both populations is negative but non-significant ($p > 0.10$) (Table 4.3; Figure 4.2e-f). No population data are available for the Gualala River, but numbers are believed to be extremely low.

Monitoring of three dependent populations in this stratum has been initiated (Table 4.4). Brush Creek has been surveyed for adult spawners for the past six seasons, but no coho salmon have been observed. Greenwood and Elk creeks have been surveyed as part of the CMP; however, these creeks are not sampled every year due to the relatively small spatial extent of potential coho salmon habitat. A small number of coho salmon redds were found in Greenwood Creek in the 2008–2009 spawning season, but they were not observed in 2012–2013. No coho salmon have been observed in Elk Creek in the two years it has been surveyed.

Coastal Stratum. Population monitoring is ongoing for two of three independent populations in the Coastal Stratum. Redd surveys have been conducted in Lagunitas Creek and its tributaries annually since the 1997–1998 spawning season by Marin Municipal Water District, the National Park Service, and the Salmon Protection and Watershed Network. Methods for expanding redd counts to adult estimates have not been developed as there is no life-cycle monitoring station in this stratum to develop spawner:redd relationships. For this assessment, we have assumed a ratio of two adults per redd (assuming one redd and one male per female). Over the 17-year period of record, the average number of adults appears to be near 500, which is approximately 20% of the recovery target of 2600 for this population (Table 4.3). The long-term trend is slightly downward, though not significant (Table 4.3; Figures 4.1g and 4.2g). Within the past six years, the population appears to have increased from a low reached in the 2008–2009 season.

Monitoring in the Russian River basin was initiated in 2003–2004 to assess the effectiveness of the hatchery program at Warm Springs. The spatial extent of sampling has increased through time as the number of streams receiving hatchery plants has grown. Likewise, methods for deriving adult estimates have also varied through time (M. Obedzinski, UC Davis, personal communication.) As a consequence, these data are not appropriate for assessing trends. However, they do provide a basis for estimating adult abundance in the Russian River watershed from the mouth to the Dry Creek watershed, inclusive, for the last four years. These estimates, which are based on a combination of information from adult traps, spawner surveys, PIT tag detections, video counts (to discriminate between fish of hatchery and natural origin), juvenile surveys, and smolt traps (to derive a minimum number of spawners in certain tributaries), indicate that population size has ranged from 206 to 536 fish, most of which are returning hatchery-origin fish. These numbers suggest the Russian River population is far below the proposed recovery target (Table 4.3)

Coho salmon were believed extirpated from the Walker Creek drainage; however, recent efforts have been made to reintroduce coho salmon to the watershed by releasing excess Olema Creek-origin adult broodstock (year 2003–2004 to 2008–2009), smolts (year 2007), and juveniles (years 2010–2014) reared at the Warm Springs Hatchery. Recent surveys have documented a total eight coho salmon carcasses and one live female during the past three spawning seasons (E. Ettlinger, MMWD, personal communication). These observations likely represent a combination of returns of hatchery smolts and natural production that has resulted from previous plantings.

Population monitoring has also been conducted by the National Park Service for two dependent populations in the stratum: Redwood Creek and Pine Gulch. As with the Lagunitas Creek surveys, no methods for expanding redd counts to adult estimates have been developed and so we have assumed a ratio of two adults per redd. Average abundance over the last 17 years has been approximately 47, which is about 17% of the recovery target of 272 (Table 4.4; Figures 4.3f and 4.4f). Coho salmon have been observed intermittently in Pine Gulch, with an average of just one adult per year over 14 years (Table 4.4; Figures 4.3e and 4.4e). Additionally, as with Walker Creek, both juvenile (years 2008) and excess broodstock adult coho salmon (years 2008–2014) have been released into Salmon Creek. These have included both Olema Creek and Russian River adults. Following the release of adults in both 2008 and 2014, juvenile coho salmon were collected from the Salmon Creek watershed, indicating successful reproduction by the released broodstock fish (M. Kittel, CDFW, personal communication).

Santa Cruz Mountain Stratum. For the last viability assessment, adult data was limited to that associated with the life-cycle monitoring station on Scott Creek. Beginning in 2012, implementation of CMP spawner surveys was initiated in the Santa Cruz Mountain Diversity Stratum in 2012 and has expanded over the past two years. However, methods for assigning unidentified redds (coho salmon vs. steelhead) have resulted in a high percentage of misassignments. Consequently, for the two independent populations in this stratum, Pescadero Creek and San Lorenzo River, expanded estimates of abundance based on redd counts are not considered reliable. In 2013–2014, a total of 19 returning jack males were collected by seine from the lower San Lorenzo River and brought to the Kingfisher Flat Hatchery in the Scott Creek watershed for use in the captive broodstock program. All of these fish were determined through coded wire tags to be hatchery fish from the Scott Creek program. In 2014–2015, three carcasses, all of hatchery origin, were recovered in Pescadero Creek, and another possible carcass was recovered in the San Lorenzo River; however, ongoing juvenile surveys (summer 2015) have not yet provided evidence of successful reproduction in either watershed. Thus both populations appear to be extirpated or nearly so.

Adult coho salmon in Scott Creek have been estimated since 2002–2003 (Figure 4.3g, 4.4g). Population abundance has averaged 71 adults over the 13 years of record (Table 4.4); however the vast majority of returning fish have been of hatchery origin, predominately 2-year old males. An estimated 163 adults (mostly hatchery fish) returned in 2014–2015 making this the largest return in a decade. This increase appears due to a combination of modified mating strategies that incorporated broodstock from Warm Springs Hatchery to combat growing concerns over inbreeding depression, coupled with implementation of a staggered release strategy, which preliminary data suggest has improved marine survival. Spawner surveys have produced only occasional observations of coho salmon in any of the dependent populations of coho salmon south of the Golden Gate over the last three seasons.

Adult coho salmon were also detected this past year in San Vicente and Waddell creeks, and subsequent summer surveys have indicated presence of juveniles in both these systems, as well as in Soquel and Laguna creeks (B. Spence and J. Kiernan, NMFS SWFSC, personal communication). Fish in most dependent populations in this stratum were considered extirpated or nearly so in the last assessment (Spence and Williams 2011).

Harvest Impacts[6]

No direct information exists on the harvest of CCC-Coho Salmon. However, it is reasonable to expect that they have a similar or more southerly distribution than Southern Oregon/Northern California Coast (SONCC)-Coho Salmon, which are primarily distributed off the coast of California and southern Oregon. Because coho salmon-directed fisheries and coho salmon retention have been prohibited off the coast of California since 1996, the CCC-Coho Salmon ocean exploitation rate is likely very low and attributable to non-retention impacts in California and Oregon Chinook-directed fisheries, non-retention impacts in Oregon mark-selective coho salmon fisheries, and impacts in Oregon non-mark selective fisheries.

The SONCC (Rogue/Klamath) natural-origin coho salmon ocean exploitation rate time series provides the best available proxy measure of trends in the CCC-Coho Salmon ocean exploitation rate. This rate has been low and relatively stable since the early 1990s (average of 5.3% for years 1994–2014), which contrasts sharply with the much higher rates estimated for the 1980s and early 1990s (average of 50.8% between 1986 and 1993) (Figure 4.5, L. LaVoy and R. Kope, NMFS, personal communication).

Freshwater fishery impacts on CCC-Coho Salmon are likely minor given California's statewide prohibition of coho salmon retention. In summary, the available information indicates that the level of CCC-Coho Salmon ESU fishery impacts has not changed appreciably since the 2010 salmon and steelhead assessment (Williams et al. 2011).

Summary and Conclusions

In summary, assessing changes in the viability of the CCC-Coho Salmon ESU remains a challenge due to the scarcity of long-term datasets for most populations. However, implementation of the CMP across significant portions of the ESU has resulted in a number of shorter time series that have substantially improved our understanding of

---

[6] Harvest impacts section prepared by Michael O'Farrell

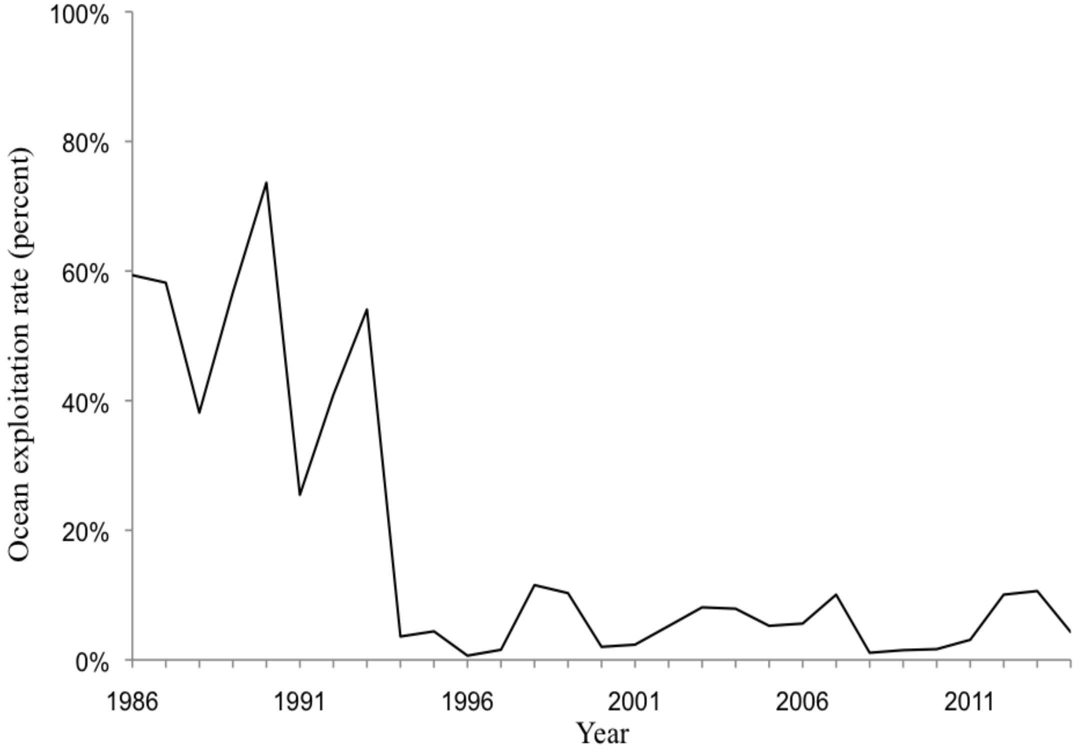

Figure 4.5. Natural-origin Rogue/Klamath coho salmon ocean exploitation rate estimates for years 1986–2014 (L. LaVoy and R. Kope, NMFS, personal communication).

current viability. The existing data indicate that all independent and dependent populations are well below recovery targets and, in some cases, exceed high-risk thresholds established by the TRTs. Although the longer-term (12–17 year) trends tend to be downward, data from the past 5 years suggest that some populations reached their lowest levels around 2008–2009 and have rebounded slightly since then. An area of particular concern is the downward trends in abundance of virtually all dependent populations across all diversity strata. These trends suggest that dependent populations are less able to maintain connectivity or act as buffers against declines in neighboring independent populations, suggesting that the independent populations are becoming more isolated with time. Populations continue to be the strongest in the Mendocino County watersheds from the Navarro River northward, and weaker to the south, with the exception of Lagunitas Creek. The viability of coho salmon in the Santa Cruz Mountain Diversity Stratum, where virtually all observed salmon have been the result of hatchery operations, remains especially dire. We conclude that the CCC-Coho Salmon ESU continues to be in danger of extinction.

4.2 California Coastal Chinook Salmon ESU

ESU Boundary Delineation

The initial status review for Chinook salmon (Myers et al. 1998) proposed a single ESU for Chinook salmon populations inhabiting coastal watersheds from Cape Blanco, Oregon, south to but not including San Francisco Bay, and including tributaries of the Klamath River downstream of its confluence with the Trinity River. Subsequent review led to division of the originally proposed ESU into the Southern Oregon and Northern California Coastal (SONCC) ESU, and the California Coastal (CC) ESU, the latter including populations spawning in coastal rivers from Redwood Creek (Humboldt County) south to the Russian River, inclusive (NMFS 1999).

The previous viability assessment (Williams et al. 2011) discussed the fact that populations that lie between the lower boundary of the Central Valley Fall-run ESU (Carquinez Straits) and the southern boundary of California Coastal Chinook Salmon ESU (Russian River) were not included in either ESU, despite the fact that Chinook salmon had been reported in several basins. Available genetic evidence indicated fish from the Guadalupe and Napa rivers in San Francisco and San Pablo bays had close affinity with Central Valley Fall-run Chinook (Garza and Pearse 2008), and it was recommended that fish from these two watersheds be included in the Central Valley Fall-run Chinook Salmon ESU. Evidence for fish in Lagunitas Creek was equivocal, with 17 samples assigned almost equally between California Coastal Chinook Salmon and Central Valley Fall-run Chinook Salmon. The review team tentatively concluded that Lagunitas Creek Chinook salmon should be considered part of the California Coastal ESU pending additional data (Williams et al. 2011). NMFS subsequently indicated that a boundary change was under consideration (76 FR 50447); however, no action has been taken to date. There is no new genetic information that helps resolve this issue (C. Garza, NMFS SWFSC, personal communication).

Summary of Previous Assessments

Myers et al. (1998) and Good et al. (2005) concluded that California Coastal Chinook salmon were likely to become endangered. Good et al. (2005) cited continued evidence of low population sizes relative to historical abundance, mixed trends in the few available time series of abundance indices available, low abundance and extirpation of populations in the southern part of the ESU, and the apparent loss of the spring-run life-history type throughout the entire ESU as significant concerns. In the most recent viability assessment, Williams et al. (2011) concluded that there was no evidence to indicate a substantial change in conditions since the previous review of Good et al. (2005). They noted that the lack of population-level estimates of adults continued to hinder assessments of status, and that although all independent populations of Chinook salmon in the North-Coastal and North Mountain Interior strata continue to persist, there is high uncertainty about the current abundance of all of these populations. Further, they cited the apparent extirpation of populations in the North-Central Coastal stratum and the loss of all but one population (Russian River) in the Central Coastal stratum as significant concerns since this gap reduced connectivity among strata across the ESU.

New Data and Updated Analyses

Abundance and Trends

At the time of the last assessment (Williams et al. 2011), population-level estimates of the abundance of Chinook salmon in this ESU were almost entirely lacking. Data were limited to time series of (1) spawner indices (maximum live/dead counts) at three sites in the Eel and Mad river basins where data have been collected since the 1970s, (2) weir counts at Freshwater Creek that began in 1994, (3) dam counts at Van Arsdale Fish Station in the upper Eel River, (4) spawner estimates for Prairie Creek, a tributary to Redwood Creek (Humboldt County), and (5) video counts of adults at Mirabel in the Russian River that began in 2000. Only the Russian River video counts likely provided some indication of total population abundance, though these counts do not include fish spawning below the counting facility. The remaining sampling efforts either provide only indices of relative abundance and not population estimates (e.g., Mad and Eel river sites), or sample only a portion of the population (e.g., Prairie Creek, Freshwater Creek, and Van Arsdale Station). Most of these sampling efforts have continued, with the exception of the Prairie Creek surveys, which were discontinued in 2012.

Since publication of the previous assessment (Williams et al. 2011), new information has become available as a result of CMP implementation in Mendocino County and portions of Humboldt County. Because some of these survey efforts have targeted coho salmon, they have not necessarily covered the full spatial and temporal extent of Chinook salmon spawning. Nevertheless, these efforts have significantly improved our understanding of the viability of Chinook salmon in this ESU. Summaries of available data are presented by diversity stratum below.

North Coastal Stratum. Population-level estimates of adult abundance for independent populations of Chinook salmon in the North-Coastal stratum remain scarce. The CMP has been implemented in Redwood Creek, Humboldt Bay tributaries, and the Mattole River for two to four years, producing estimates of the total number of Chinook salmon redds in these watersheds (Table 4.5). However, to date, methods for expanding redd counts to population estimates have not yet been developed (S. Ricker, CDFW, personal communication). Additionally, sampling generally targets the spawning period and habitat for coho salmon and thus may not encompass the entirety of the spawning period and space for Chinook salmon (Ricker et al. 2014d; Ricker and Anderson 2014; Ricker et al. 2015h). With these caveats in mind, the data indicate that Redwood Creek has produced 921 Chinook salmon redds annually (range 752–1042) over the last four years. The average redd estimate for the Mattole River for the past two seasons was 250 (range 128–373). The Humboldt Bay tributaries produced an average of only three Chinook salmon redds (range 0–13) over the past 4 seasons (Table 4.5). Without methods for expanding redd counts to adult estimates, these numbers cannot be directly compared to

Table 4.5. Viability metrics for independent populations of Chinook salmon in the CC-Chinook Salmon ESU. NA indicates not available or applicable. Trends are shown only for populations where time series is at least six years, **bold** indicates significant trend. IPkm includes only habitats that are currently accessible. $N_{a(arith)}$ target refers to the low-risk viability target identified by the Technical Recovery Team (Spence et al. 2008).

Stratum/population	Years	$\overline{N}_{a(arith)}$	$\overline{N}_{a(geom)}$	$\overline{N}_{g(harm)}$	$\hat{C}$	$\hat{T}$ *(95% CI)*	IPkm	$\hat{D}_{dep}$	$\hat{D}_{ssd}$	$\hat{D}_{ssd}$ target	$\overline{N}_{a(arith)}$ target
North Coastal											
Redwood Creek[a]	4	921	915	2824	NA	NA	116.1	7.8	7.9	29.3	3400
Little River	-	-	-	-	-	-	18.6	-	-	40.0	700
Mad River	-	-	-	-	-	-	94.0	-	-	31.8	3000
Humboldt Bay[a]	4	3	2	0	NA	NA	76.7	0.0	0.0	33.7	2600
Lower Eel River	-	-	-	-	-	-	514.9	-	-	20.0	10300
Bear River	-	-	-	-	-	-	39.4	-	-	37.8	1500
Mattole River[1]	2	250	219	-	NA	NA	177.5	-	-	22.5	4000
North Mtn. Interior											
Lower Eel River	-	-	-	-	-	-	-	-	-	-	-
Upper Eel River	-	-	-	-	-	-	495.3	-	-	20.0	11100
North-Central Coastal											
Ten Mile River[b]	6	14	5	51	NA	-0.215 (-1.520, 1.091)	67.2	0.1	0.2	34.8	2300
Noyo River[b]	6	13	8	24	NA	**-0.624 (-0.951, -0.296)**	62.2	0.1	0.2	35.3	2200
Big River[b]	6	15	8	33	NA	-0.588 (-1.476, 0.300)	104.3	0.1	0.1	30.6	3200
Central Coastal											
Navarro River[b]	6	3	2	0	NA	-0.274 (-1.110, 0.562)	131.5	-	-	27.6	3600
Garcia River[b]	6	5	3	13	NA	0.048 (-0.888, 0.983)	56.2	0.1	0.1	36.0	2000
Gualala River[b]	-	-	-	-	-	-	175.6	-	-	22.7	4000
Russian River[c]	14	3257	2806	8664	0.67	0.019 (-0.067, 0.104)	496.4	6.1	2.8	20.0	11700

a – Numbers indicate the estimated number of redds in the population (expanded from counts).
b – Numbers indicate the estimated number of adults based on fish/redd expansions from life-cycle monitoring stations.
c – Numbers are based on video counts at Mirabel Dam; a small but unknown percentage of adults spawn below this location, so the estimate does not include entire population.

viability targets; however, it is evident that none of these three populations are approaching viability targets at this time.

Besides these population level estimates, longer time series of partial population estimates or index reach maximum live/dead counts are available for Prairie Creek (part of the Redwood Creek population), Cannon Creek (part of the Mad River population), Freshwater Creek (part of the Humboldt Bay population), the South Fork Eel River (part of the Lower Eel River population), and Sproul Creek (part of the Lower Eel River Population) (Table 4.6). The Prairie Creek time series showed an average of 272 adult Chinook salmon (range 38–710) in this subwatershed over the 14-year period of record, with a significant ($p = 0.015$) negative trend at the time the survey was discontinued (Table 4.6; Figures 4.6a, 4.7a). Spawner surveys have been performed on Cannon Creek since 1981, with data reported as maximum live/dead counts (Table 4.6). The 34-year trend for this dataset has been positive, but not significantly so ($p = 0.212$), while the 16-yr trend has been negative but not significant ($p = 0.235$) (Table 4.6; Figure 4.6b, 4.7b). Counts of Chinook salmon have been made at a weir on Freshwater Creek since 2001 (Ricker 2015); these counts are partial counts as fish can pass over the weir during periods of high flow and smaller jacks may pass through the weir. On average, 21 natural-origin adults[7] have been counted annually over the 15-year period of record. The trend over this period has been negative and significant ($p < 0.001$; Table 4.6; Figures 4.6c, 4.7c). Estimates of Chinook salmon redds have been made four last four years in the South Fork Eel River (Ricker et al. 2015a-d); the average estimate has been 772 (range 149–1345) during this period (Table 4.6). Finally, spawner surveys have been performed on Sproul Creek since 1975, with data reported as maximum live/dead counts. The 39-year trend for this dataset has been negative but not significant ($p = 0.150$), whereas the more recent 16-year trend has been positive but also not significant ($p = 0.453$) (Table 4.6; Figures 4.6d, 4.7d).

North Mountain Interior Stratum. The North Mountain Interior stratum contains the upper Eel River Chinook salmon population, as well as the portion of the lower Eel River population that inhabits watersheds of the interior mountains of the Eel River basin, including the Van Duzen River and Larabee Creek basins. For the upper Eel River population, there are no population-level estimates of abundance available. However, two time series of partial abundance data are available: maximum live/dead counts for an index reach in Tomki Creek (since 1976) and weir counts at Van Arsdale Station (since 1947). Counts at both of these locations appear highly influenced by flow conditions in the mainstem, which in turn are affected by water releases from Cape Horn and Scott dams. In years of low flow, fish appear less inclined to enter Tomki Creek or ascend the Eel River as far as Van Arsdale Station and instead spawn in areas downstream; thus, the reliability of these counts as indices of abundance is somewhat questionable (S. Harris, CDFW, personal communication). Beginning in 2004, mandated increases in minimum flow releases from Cape Horn Dam have been implemented (NMFS 2002; J. Jahn, Table

[7] A small hatchery program for Chinook salmon on Freshwater Creek was discontinued in the early 2000s.

4.6. Population information for CC-Chinook salmon populations with only index data or partial population estimates. NA indicates not available or applicable. Trends are shown only for populations where time series is at least six years, **bold** indicates significant trend. Short-term (16-yr) trends are shown along with long-term trends for those datasets spanning 30 or more years.

Stratum/population	Years	$\overline{N}_{a(arith)}$	$\overline{N}_{a(geom)}$	$\overline{N}_{g(harm)}$	$\hat{T}$ *(95% CI)*
North Coastal					
Prairie Creek[a]	14	272	190	436	**-0.140 (-0.248, -0.032)**
Cannon Creek[b]	34	102	61	161	0.027 (-0.016, 0.069)
	16	122	92	355	-0.054 (-0.147, 0.039)
Freshwater Creek[c]	15	21	8	16	**-0.240 (-0.349, -0.130)**
SF Eel River[d]	4	772	585	2190	NA
Sproul Creek[e]	39	226	125	394	-0.025 (-0.060, 0.010)
	16	145	100	398	0.043 (-0.077. 0.453)
North Mtn. Interior					
Tomki Creek[f]	34	554	104	150	**-0.100 (-0.152, -0.048)**
	16	78	48	210	0.013 (-0.125, 0.151)
Van Arsdale Station[g]	63	370	40	21	**0.078 (0.049, 0.108)**
	16	906	608	1340	0.087 (-0.004, 0.179)

a – Prairie Creek represents a portion of the Redwood Creek population. Numbers are population estimates based on Area-under-the-curve (AUC) method. Surveys were discontinued when basin-scale monitoring of Redwood Creek was initiated in 2012.
b – Cannon Creek is an index reach in the Mad River basin. Numbers are maximum live/dead counts. Survey effort varies annually.
c – Freshwater Creek represents a portion of the Humboldt Bay population. Numbers are weir counts of natural-origin fish; populations were too small to develop reliable population estimates.
d – SF Eel River represents a portion of the Lower Eel River population. Numbers are expanded estimates of the number of redds; they are not population estimates.
e – Sproul Creek represents a portion of the Lower Eel River population. Numbers are maximum live/dead counts. Survey effort varies annually.
f – Tomki Creek represents a portion of the Upper Eel River population. Numbers are maximum live/dead counts. Survey effort varies annually.
g – Van Arsdale Station counts represent a portion of the Upper Eel River population. Numbers are counts of fish passed over the dam and represent a variable fraction of the total population, as the proportion of individuals reaching the dam appears highly flow dependent. Values for the last 16 years are based on naturally produced fish only; hatchery fish were excluded.

NMFS Southwest Region, personal communication), resulting in a general increase in the amount of water available in the mainstem Eel River below the dam. The increase in flow has likely influenced the distribution of spawners in the Eel River, possibly drawing more fish as far as Van Arsdale Station. With these caveats in mind, maximum live/dead counts in Tomki Creek have averaged 554 (range 3–3,666) over the 34-year period of record, but only 78 (range 5–226) over the last 16 years (Table 4.6). The long-term trend in these counts is negative ($p < 0.001$); however, the short-term trend has been positive though marginally significant ($p = 0.060$), primarily because of three relatively strong years in succession from 2010–2011 to 2012–2013 (Figures 4.6e, 4.7e). Counts at Van Arsdale are also confounded by the fact that between 1996 and 2004, an average of

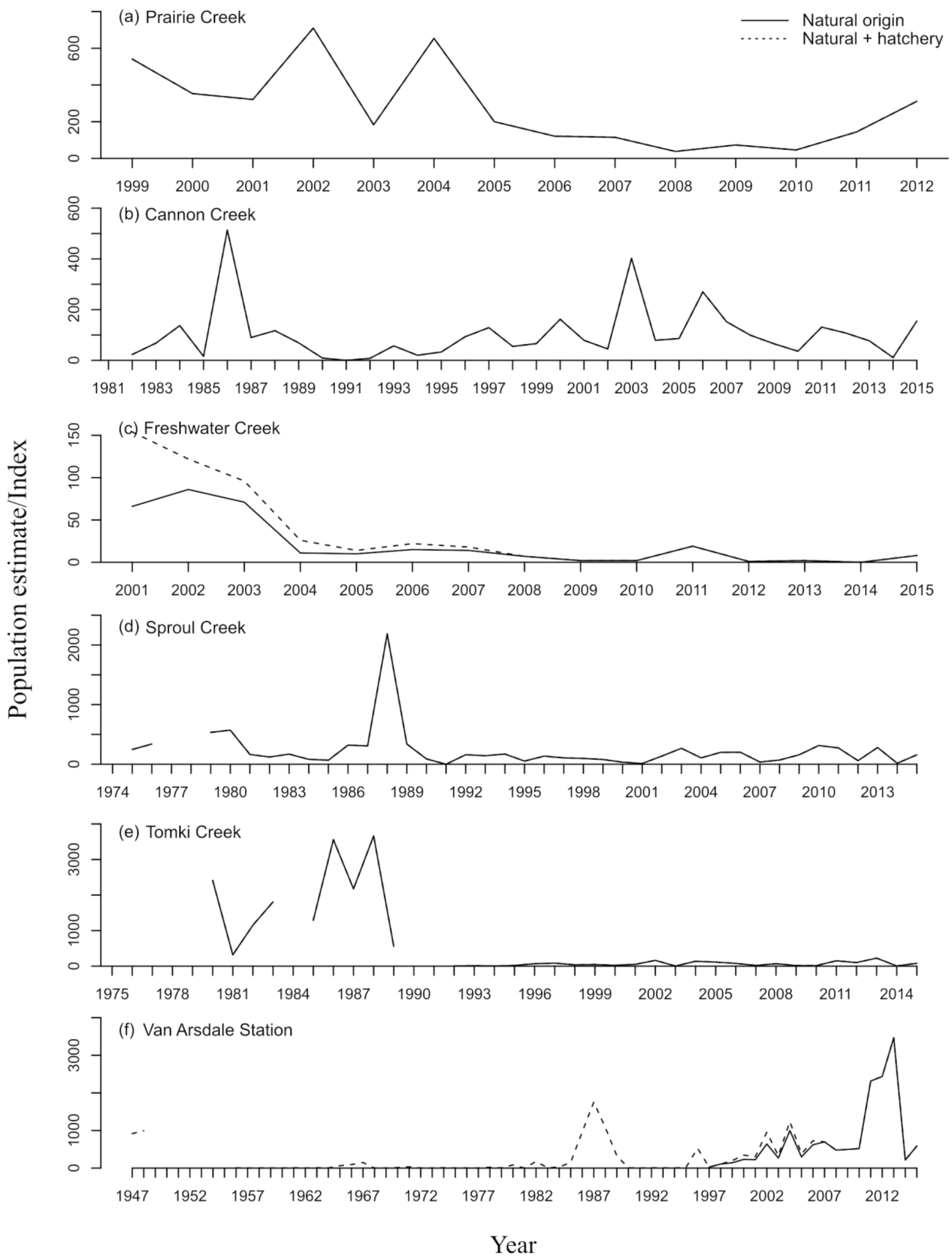

Figure 4.6. Time series of population indexes or partial population estimates for independent populations of CC-Chinook Salmon. Values for Cannon, Sproul, and Tomki creeks are maximum live-dead indexes. Van Arsdale Station and Freshwater Creek are weir counts. Prairie Creek is based on area-under-the-curve (AUC) estimates for the watershed.

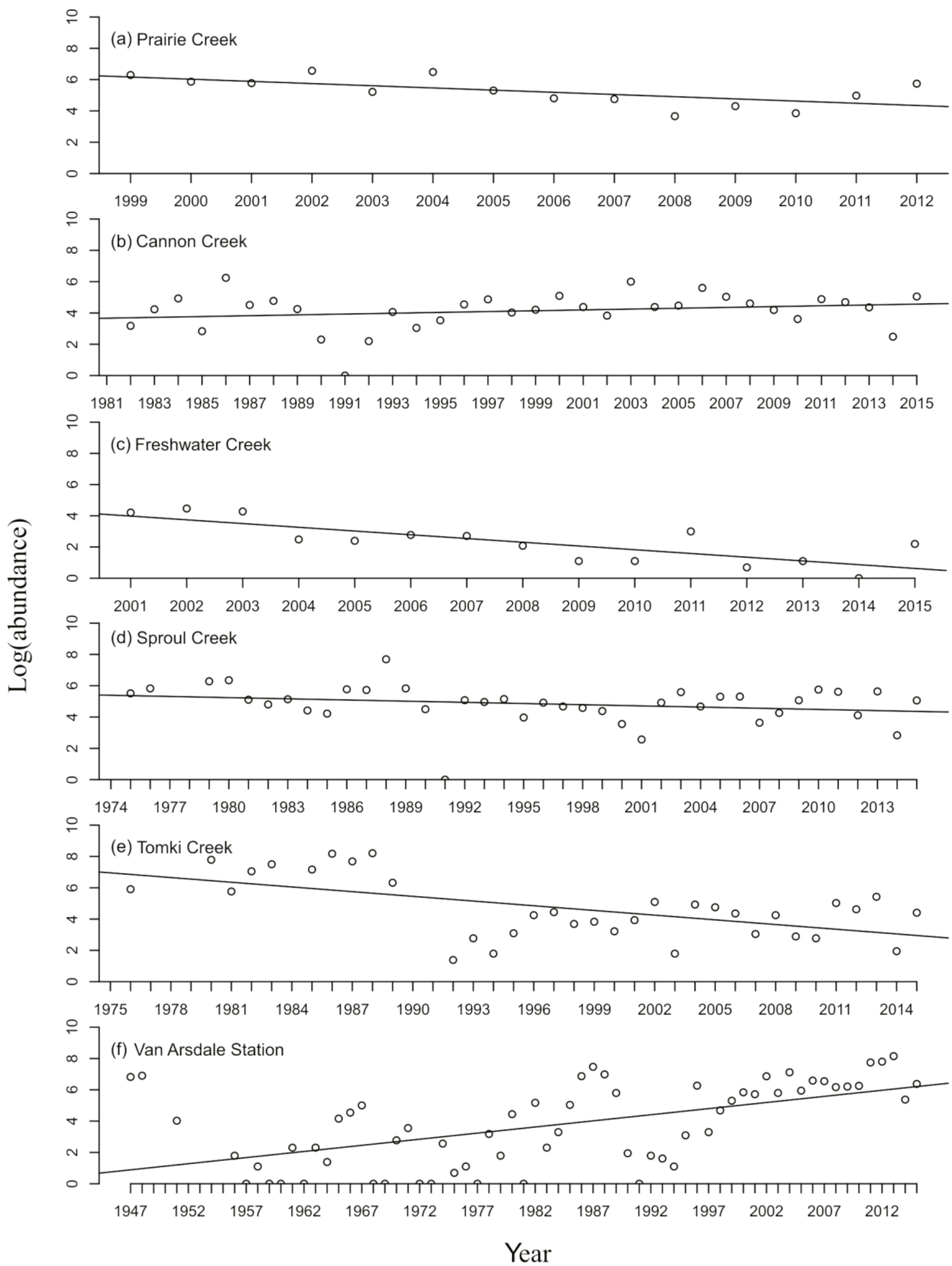

Figure 4.7. Population trends (log abundance) for indexes or partial population estimates for independent populations of CC-Chinook Salmon. Values for Cannon, Sproul, and Tomki creeks are based on maximum live-dead indexes. Van Arsdale Station and Freshwater Creek are weir counts. Prairie Creek is based on area-under-the-curve (AUC) estimates for watershed.

38,822 hatchery Chinook salmon were released into the Eel River annually. Over the last 16 years, counts of natural-origin adults have averaged 906 (range 215–3,446), and there has been a significant positive trend (Table 4.6; Figures 4.6f, 4.7f)). However, although trends were calculated based only on natural origin fish, an unknown proportion of these fish are likely recent descendants of hatchery-origin fish. Thus, it is unclear whether the recent positive trend reflects increases in wild spawners, redistribution of fish associated with changes in flow releases from upstream dams, or legacy effects of past hatchery plantings.

In addition to these longer time series of abundance information, attempts have also been made to conduct spawner surveys in the mainstem Eel River as well as several major tributaries, including the Middle Fork Eel River, Outlet Creek, and Tomki Creek. For the 2013–2014 spawning season, these efforts produced an estimate of 3,152 adult Chinook salmon, inclusive of fish captured at Van Arsdale Station (Harris and Thompson 2014). A similar effort in the 2009–2010 spawning season produced an estimate of approximately 3,500 fish for portions of the mainstem Eel River, Tomki Creek, Outlet Creek and one of its tributaries, and Van Arsdale Station (Harris 2010). Attempts to estimate Chinook salmon abundance in 2012–2013 were unsuccessful due to significant rains that resulted in poor survey conditions (Harris and Thompson 2013). Nevertheless, these data indicate that the Van Arsdale and Tomki Creek estimates constitute only a relatively small fraction of the total Upper Eel River Chinook salmon population.

North-Central Coastal Stratum. The previous viability assessment noted the apparent extirpation of Chinook salmon populations in watersheds of the North-Central Coastal Stratum (Williams et al. 2011). Implementation of the CMP throughout this stratum beginning in 2009 has produced data that indicate this is not true. Estimates based on expanded redd counts indicate that the Ten Mile, Noyo, and Big rivers continue to produce small numbers of Chinook salmon in most years, with each of these watersheds averaging 13–15 fish per year over the last six years (Table 4.5; Figures 4.8a-c, 4.9a-c). Although in all cases these numbers are less than 1% of the viability targets and fall below the depensation thresholds for high risk, they nevertheless provide evidence that Chinook salmon are still regularly using these watersheds to spawn.

Central Coastal Stratum. Population monitoring is currently occurring for three of four independent populations of Chinook salmon in the Central Coastal Stratum. Monitoring of the Navarro and Garcia river populations was initiated in 2009. This monitoring has confirmed presence of very low numbers of Chinook salmon, with estimates averaging 3 and 5 adults for these two watersheds, respectively, in the past six years (Table 4.5; Figures 4.8d-e, 4.9d-e). Monitoring of adult Chinook salmon using video counts at Mirabel Dam on the Russian River has been conducted since 2001. An average of 3,257 Chinook salmon have been counted annually over the 14-year period of record and there has been essentially no trend in abundance ($p = 0.644$) (Table 4.5; Figures 4.8f, 4.9f). The average count represents about 28% of the viability target for the Russian River; however, some spawning by Chinook salmon does occur below Mirabel Dam, so the population is likely closer to the target than these numbers indicate.

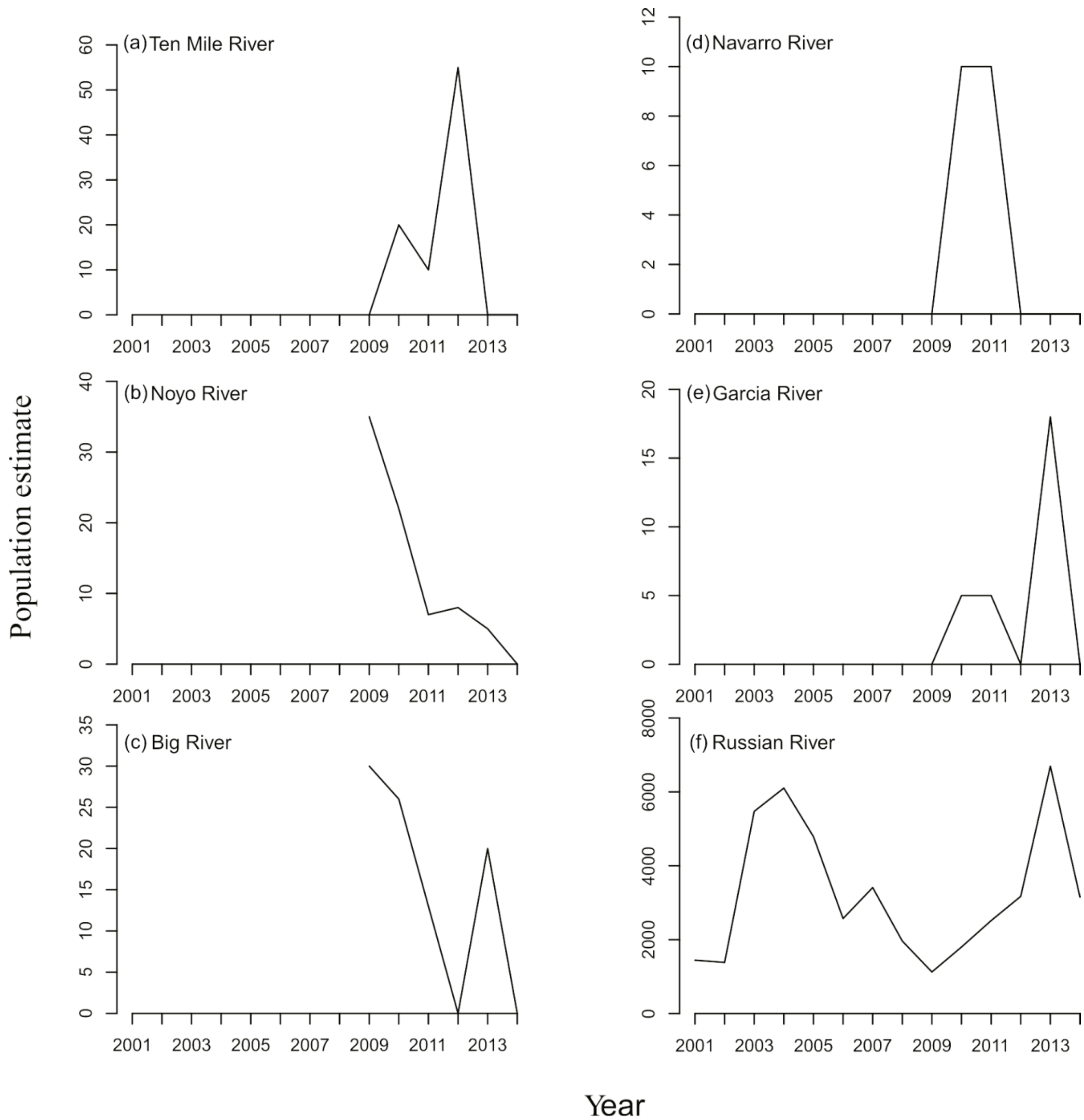

Figure 4.8. Time series of population abundance estimates for independent populations of CC-Chinook Salmon. Values for Russian River are video counts at Mirabel Dam. All other estimates are based on fish/redd expansions from life-cycle monitoring stations.

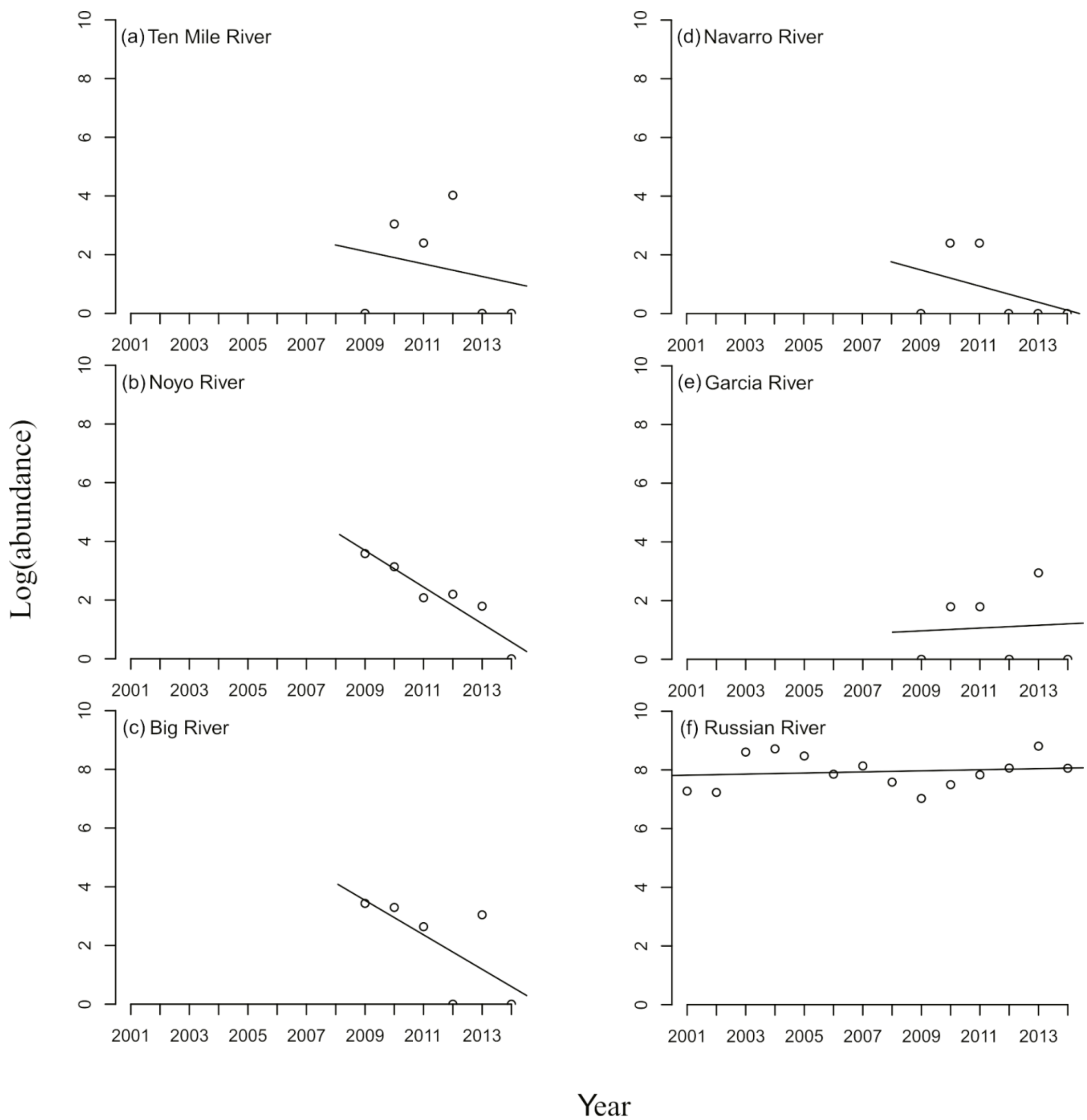

Figure 4.9. Population trends (log abundance) for independent populations of CC-Chinook Salmon. Values for Russian River are video counts at Mirabel Dam. All other estimates are based on fish/redd expansions from life-cycle monitoring stations.

Harvest Impacts[8]

Very limited data exits on the harvest of California Coastal Chinook Salmon (CC-Chinook Salmon). Owing to this data deficiency, the Klamath River Fall-run Chinook salmon (KRFC) age-4 (fully vulnerable) ocean harvest rate is used as a fishery management proxy to limit harvest impacts on CC-Chinook Salmon. The CC-Chinook Salmon ocean fishery consultation standard is a maximum predicted KRFC age-4 ocean harvest rate of 16%.

The KRFC age-4 ocean harvest rate fell sharply from its average value of 44% over the period 1981–1990 to estimates that have largely remained below 20% since 1991. Very low KRFC age-4 ocean harvest rates were observed between 2008 and 2012, partially reflecting the widespread fishery closures in California and Oregon between 2008 and 2010 (Figure 4.10). The average KRFC age-4 ocean harvest rate estimated over the years since the last viability assessment (2011–2014) is 13%, which falls below the 16% CC-Chinook salmon consultation standard.

Freshwater fishery impacts on CC-Chinook Salmon are likely relatively minor because retention of Chinook salmon is prohibited.

In summary, the available information indicates that the level of CC-Chinook Salmon fishery impacts has not changed appreciably since the 2010 salmon and steelhead assessment (Williams et al. 2011).

Summary and Conclusions

The lack of long-term population-level estimates of abundance for Chinook salmon populations in the CC ESU continues to hinder viability assessment, though the situation has improved with implementation of the CMP in the Mendocino Coast Region and portions of Humboldt County. The available data, a mixture of short-term (6-year or less) population estimates or expanded redd estimates and longer-term partial population estimates and spawner/redd indexes, provide no indication that any of the independent population are approaching viability targets. However, there remains high uncertainty regarding key populations, including the Upper and Lower Eel River populations and the Mad River population, due to incomplete monitoring across the spawning habitat of Chinook salmon in these basins (O'Farrell et al. 2012). Because of the short duration of most time series for independent populations, little can be concluded from trend information. The longest time series, video counts in the Russian River, indicates that the population has remained fairly steady of the 14-year period of record. The longer time series associated with index reaches or partial populations suggest mixed patterns, with some showing significant negative trends (Prairie Creek, Freshwater Creek, Tomki Creek), one showing a significant positive trend (Van Arsdale Station), and the remainder no significant trends. Overall, there is a lack of compelling evidence to suggest that the viability of these populations has improved or deteriorated appreciably since the previous assessment (Williams et al. 2011)

[8] Harvest impact section prepared by Michael O'Farrell

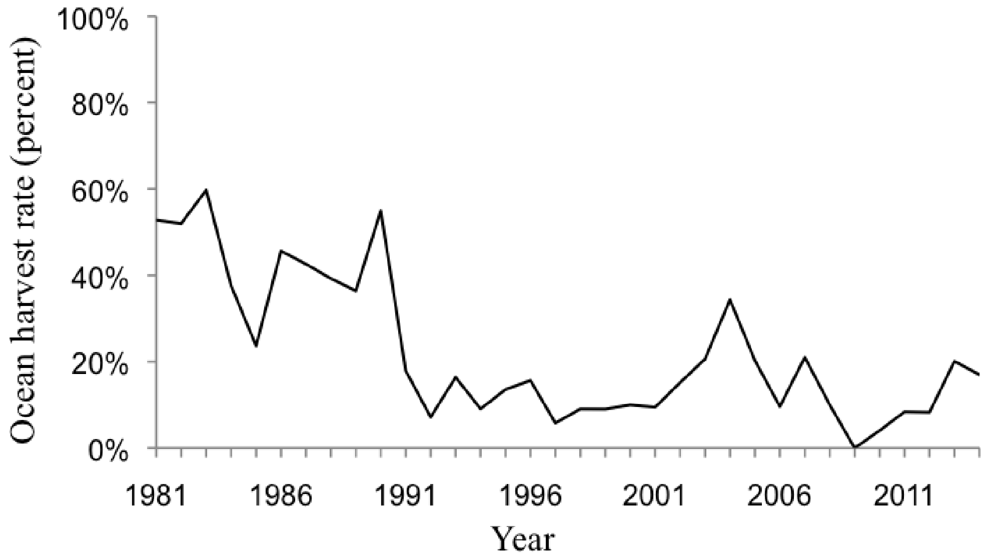

Figure 4.10. Klamath River Fall-run Chinook salmon age-4 ocean harvest rate for years 1981–2014 (PFMC 2015a).

At the ESU level, the loss of the spring-run life-history type represents a significant loss of diversity within the ESU, as has been noted in previous status reviews and viability assessments (Good et al. 2005; Williams et al. 2011). Concern remains about the extremely low numbers of Chinook salmon in most populations of the North-Central Coast and Central Coastal Diversity strata, which diminishes connectivity across the ESU. However, the fact that Chinook salmon have regularly been reported in the Ten Mile, Noyo, Big, Navarro, and Garcia rivers represents a significant improvement in our understanding of the viability of these populations in watersheds where they were thought to have been extirpated. These observations suggest that spatial gaps between extant populations are not as extensive as previously believed. In summary, the new information available since the last assessment (Williams et al. 2011) does not appear to suggest there has been a change in extinction risk for this ESU.

4.3 Northern California Steelhead ESU/DPS

DPS Boundary Delineation

See discussion of steelhead DPS boundary issues in introduction.

Summary of Previous Assessments

Busby et al. (1996) and Good et al. (2005) concluded that the Northern California (NC) Steelhead ESU/DPS was not presently in danger of extinction, but was likely to become endangered in the foreseeable future. Concerns raised by both of these biological review teams included low population abundance relative to historical estimates, recent downward trends in most stocks for which data were available, and the low abundance of summer steelhead populations. They also cited continued habitat degradation, the increasing abundance of a nonnative predator (Sacramento pikeminnow, *Ptychocheilus grandis*) in the Eel River, the influence of artificial propagation on certain wild populations, and the lack of data for this DPS as concerns and sources of risk (Busby et al. 1996; Good et al. 2005). In the most recent assessment, Williams et al. (2011) concluded that there was little evidence to indicate that the viability of the NC-Steelhead DPS had changed appreciably in either direction since publication of the previous status review (Good et al. 2005). They noted that the assessment was hindered by the scarcity of population-level estimates of abundance for either winter- or summer-run populations within this DPS. The available information suggested mixed trends in abundance, with more populations showing decreases than increases. However, they suggested that these declines were likely the result of a combination of drought conditions that prevailed between 2007 and 2009 coupled with apparent poor ocean conditions (Williams et al. 2011).

New Data and Updated Analyses

<u>*Abundance and Trends*</u>

At the time of the last assessment, population-level estimates of abundance were available for less than 10% of independent populations of winter- and summer-run steelhead within the DPS (Williams et al. 2011). Since that time, the CMP has been more broadly implemented in Mendocino County as well as selected watersheds in Humboldt County. Data from the CMP are now available for 17 independent populations, as well as six dependent populations or partial populations (most associated with life-cycle monitoring stations). The majority of these datasets span a period of six or fewer years; however, they do provide the first comprehensive estimates of adult abundance or redds for a number of populations. Significant data gaps do remain, however, particularly in the Lower Interior and North Mountain Interior diversity strata, which encompass most of the Eel River populations, excluding the South Fork Eel River. In addition to these newer datasets, several longer time series of adult abundance for partial populations remain available, though in two instances, these monitoring efforts have been discontinued. Summaries of available data are presented below by diversity stratum.

Northern Coastal Stratum. Implementation of the CMP for winter-run steelhead has been initiated in four watersheds in the Northern Coastal Stratum: Redwood Creek, Humboldt Bay, the South Fork Eel River, and Mattole River. These efforts have produced estimates of total redd numbers in each of these waters for the past 2–4 years (Table 4.7). However, methods for expanding redd counts to population estimates have not yet been developed (S. Ricker, CDFW, personal communication). Additionally, sampling targets the spawning period and habitat for coho salmon and thus may not encompass the entirety of the spawning period and space for steelhead (Ricker et al. 2014d, 2015d, 2015h). With these caveats in mind, the average steelhead redd estimate for Redwood Creek has been 154 (range 52–389) over the last four years. The average redd estimate for Humboldt Bay over the same period has been 88 (range 17–183). For the South Fork Eel River, redd counts have averaged 643 (range 352–1113) over the last four years. Only two years of data are available for the Mattole River, with an average steelhead redd estimate of 298 (range 194–402). Because surveys do not encompass the entire spawning period in some years and methods have not been developed for expanding redd estimates to adult abundance estimates, these numbers cannot be directly compared to viability targets. Nevertheless, it appears evident that all four of these populations are well below viability targets (Table 4.7).

A longer time series of adult abundance estimates is available for Prairie Creek (14 years), although this monitoring effort was recently discontinued (2012) and replaced with the CMP effort that spans the entire Redwood Creek watershed. These surveys produced estimates averaging 40 spawners annually, with a slight positive but nonsignificant trend ($p = 0.545$) (Table 4.8; Figure 4.11a, 4.12a). Estimates of steelhead abundance in Freshwater Creek have been generated using mark-recapture methods since 1999. Over this 15-year period, an estimated average of 170 steelhead (range 51–432) have returned to Freshwater Creek annually, and the trend has been negative but not significantly so ($p = 0.108$) (Table 4.8; Figures 4.11b, 4.12b). Information is not available for the Maple Creek/Big Lagoon, Little River, Mad River, Price Creek, or Bear River winter-run steelhead populations.

Information on the abundance of summer-run steelhead populations is collected in two systems in the Northern Coastal Stratum: Redwood Creek and the Mattole River. Dive surveys covering an index reach of approximately 25.9 km of Redwood Creek have been conducted annually since 1981. Mean counts have averaged only 10 fish during the period of record (range 0–44), during which there has been a negative but non-significant ($p = 0.720$) trend (Table 4.9; Figures 4.13a, 4.14a). The recent (16-year) trend has been positive and marginally significant ($p = 0.077$); however, the population remains at critically low abundance. Dive counts of summer steelhead have also been made annually on the Mattole River since 1996 by the Mattole Salmon Group. Over this 19-year period, an average of 73 fish (range 35–129) have been observed annually (Table 4.9; Figures 4.13c, 4.14c), with about 33% being adults and the remaining 67% half-pounders (MSG 2015). Because the spatial extent of the survey has varied among years, analysis of trends was deemed inappropriate. Summer dive surveys were conducted on the Mad River

Table 4.7. Viability metrics for independent winter-run populations of steelhead in the NC-Steelhead DPS. NA indicates not available or applicable. Trends shown only for populations where time series is at least six years, **bold** indicates significant trend. IPkm includes only habitats that are currently accessible. $N_{a(arith)}$ target refers to the low-risk viability target identified by the Technical Recovery Team (Spence et al. 2008).

Stratum/population	Years	$\overline{N}_{a(arith)}$	$\overline{N}_{a(geom)}$	$\overline{N}_{g(harm)}$	$\hat{C}$	$\hat{T}$ (95% CI)	IPkm	$\hat{D}_{dep}$	$\hat{D}_{ssd}$	$\hat{D}_{ssd}$ target	$\overline{N}_{a(arith)}$ target
Northern Coastal											
Redwood Creek[a]	4	154	112	529	NA	NA	270.9	NA	NA	20.0	5400
Maple Cr/Big Lagoon	-	-	-	-	-	-	71.7	-	-	32.3	2300
Little River	-	-	-	-	-	-	63.0	-	-	33.5	2100
Mad River	-	-	-	-	-	-	453.7	-	-	20.0	5800
Humboldt Bay[a]	4	88	62	283	NA	NA	212.1	NA	NA	20.0	4200
Price Creek	-	-	-	-	-	-	18.2	-	-	39.7	700
SF Eel River[a]	4	643	574	1752	NA	NA	1017.0	NA	NA	20.0	20300
Bear River	-	-	-	-	-	-	107.8	-	-	27.2	2900
Mattole River[a]	2	298	279	NA	NA	NA	541.1	NA	NA	20.0	10800
Lower Interior											
Jewett Creek	-	-	-	-	-	-	16.8	-	-	39.9	700
Pipe Creek	-	-	-	-	-	-	17.4	-	-	39.8	700
Chamise Creek	-	-	-	-	-	-	36.2	-	-	37.2	1300
Bell Springs Creek	-	-	-	-	-	-	18.1	-	-	39.7	700
Woodman Creek	-	-	-	-	-	-	35.0	-	-	37.4	1300
Outlet Creek	-	-	-	-	-	-	192.6	-	-	20.0	3500
Tomki Creek	-	-	-	-	-	-	90.8	-	-	29.6	2700
Bucknell Creek	-	-	-	-	-	-	19.1	-	-	39.6	800
North Mtn. Interior											
Van Duzen River	-	-	-	-	-	-	317.4	-	-	20.0	6300

Table 4.7. continued.

Stratum/population	Years	$\overline{N}_{a(arith)}$	$\overline{N}_{a(geom)}$	$\overline{N}_{g(harm)}$	$\hat{C}$	$\hat{T}$ *(95% CI)*	IPkm	$\hat{D}_{dep}$	$\hat{D}_{ssd}$	$\hat{D}_{ssd}$ target	$\overline{N}_{a(arith)}$ target
Larabee Creek	-	-	-	-	-	-	88.4	-	-	29.9	2600
Dobbyn Creek	-	-	-	-	-	-	49.1	-	-	35.4	1700
Kekawaka Creek	-	-	-	-	-	-	30.7	-	-	38.0	1200
NF Eel River	-	-	-	-	-	-	318.2	-	-	20.0	6400
MF Eel River	-	-	-	-	-	-	503.5	-	-	20.0	10000
Upper Mainstem Eel R.	-	-	-	-	-	-		-	-	NA	NA
North-Central Coastal											
Usal Creek[b]	6	61	42	201	NA	0.366 (-0.271, 1.002)	17.6	2.3	3.5	39.8	700
Cottaneva Creek[b]	4	77	28	NA	NA	NA	23.2	NA	3.3	39.0	900
Wages Creek[b]	4	63	33	226	NA	NA	17.7	4.3	3.6	39.8	700
Ten Mile River[b]	6	407	153	893	NA	1.069 (-0.084, 2.222)	181.3	0.8	2.2	20.0	3600
Pudding Creek[b]	13	100	66	165	0.91	**-0.170 (-0.305, -0.034)**	24.3	0.7	4.0	38.9	900
Noyo River[b]	13	343	307	995	0.25	0.027 (-0.047, 0.101)	157.6	1.6	2.2	20.0	3200
Big River[b]	6	633	323	838	NA	**0.714 (0.435, 0.993)**	256.1	0.4	2.5	20.0	5100
Albion River[b]	6	60	37	104	NA	**0.457 (0.023, 0.892)**	48.6	0.3	1.2	35.5	1700
Big Salmon Creek[b]	3	91	41	NA	NA	NA	18.3	NA	5.0	39.7	700
Central Coastal											
Navarro River[b]	6	366	302	890	NA	**0.338 (0.099, 0.577)**	379.9	0.5	0.9	20.0	8000
Elk Creek[b]	2	31	13	NA	NA	NA	21.5	NA	1.4	39.2	800
Brush Creek[b]	6	13	6	22	NA	0.421 (-0.574, 1.417)	23.8	0.1	0.5	38.9	900
Garcia River[b]	6	326	258	1127	NA	0.193 (-0.332, 0.717)	137.2	2.1	2.4	23.2	3200
Gualala River	-	-	-	-	-	-	400.4	-	-	20.0	8000

a – Numbers indicate the estimated number of redds in the population (expanded from counts).
b – Numbers indicate the estimated number of adults based on fish/redd expansions from life-cycle monitoring stations.

Table 4.8. Population information for dependent populations of winter-run NC-Steelhead or populations with only index data or partial population estimates. NA indicates not available or applicable. Trends are shown only for populations where time series is at least six years, **bold** indicates significant trend.

Stratum/population	Years	$\overline{N}_{a(arith)}$	$\overline{N}_{a(geom)}$	$\overline{N}_{g(harm)}$	$\hat{T}$ *(95% CI)*
Northern Coastal					
Prairie Creek[a]	14	40	20	52	0.051 (-0.126, 0.227)
Freshwater Creek[b]	15	170	146	446	-0.055 (-0.124, 0.014)
North Mtn. Interior					
Van Arsdale Station[c]	78	1854	931	2157	**-0.033 (-0.043, -0.022)**
	16	328	278	832	**0.068 (0.011, 0.125)**
North-Central Coastal					
SF Noyo River[b]	15	81	72	233	0.018 (-0.052, 0.087)
Hare Creek[b]	9	51	14	257	**-0.451 (-0.686, -0.215)**
Caspar Creek[b]	13	54	37	122	-0.113 (-0.253, 0.027)
Little River[b]	13	18	13	41	-0.092 (-0.212, 0.028)
Central Coastal					
NF Navarro River[b]	2	358	342	NA	NA
Greenwood Creek[b]	2	7	4	NA	NA
Wheatfield Fk Gualala River[d]	9	1735	1163	5271	-0.102 (-0.407, 0.202)

a – Numbers based on AUC estimates. Surveys were discontinued after 2012 when basin-wide surveys for Redwood Creek were initiated.
b – Numbers indicate the estimated number of adults based on fish/redd expansions from life-cycle monitoring stations.
c – Numbers based on fish counts at Van Arsdale Station. Represents a partial composite of Upper Eel River and Soda Creek populations. Statistics on 78-year time series include an unknown number of hatchery-origin fish; recent (16-year) statistics are for natural-origin fish only.
d – Numbers based on observations of live fish during boat surveys. Surveys were discontinued after 2010.

between 1980 and 2005 when the effort was discontinued. However, beginning in 2012, snorkel surveys were re-initiated with the goals of implementing consistent protocols and covering the river from Kadle Hole (near Hwy 101) to R.W. Matthews Dam. Over this three-year period, an average of 427 summer steelhead have been counted annually (range 308–558), with adults constituting 73% of fish counted and half-pounders the remaining 27% (Pounds et al. 2015; D. Feral, Mad River Alliance, personal communication). Assuming the last few years are representative of current viability, the population is at roughly half its viability target (Table 4.9).

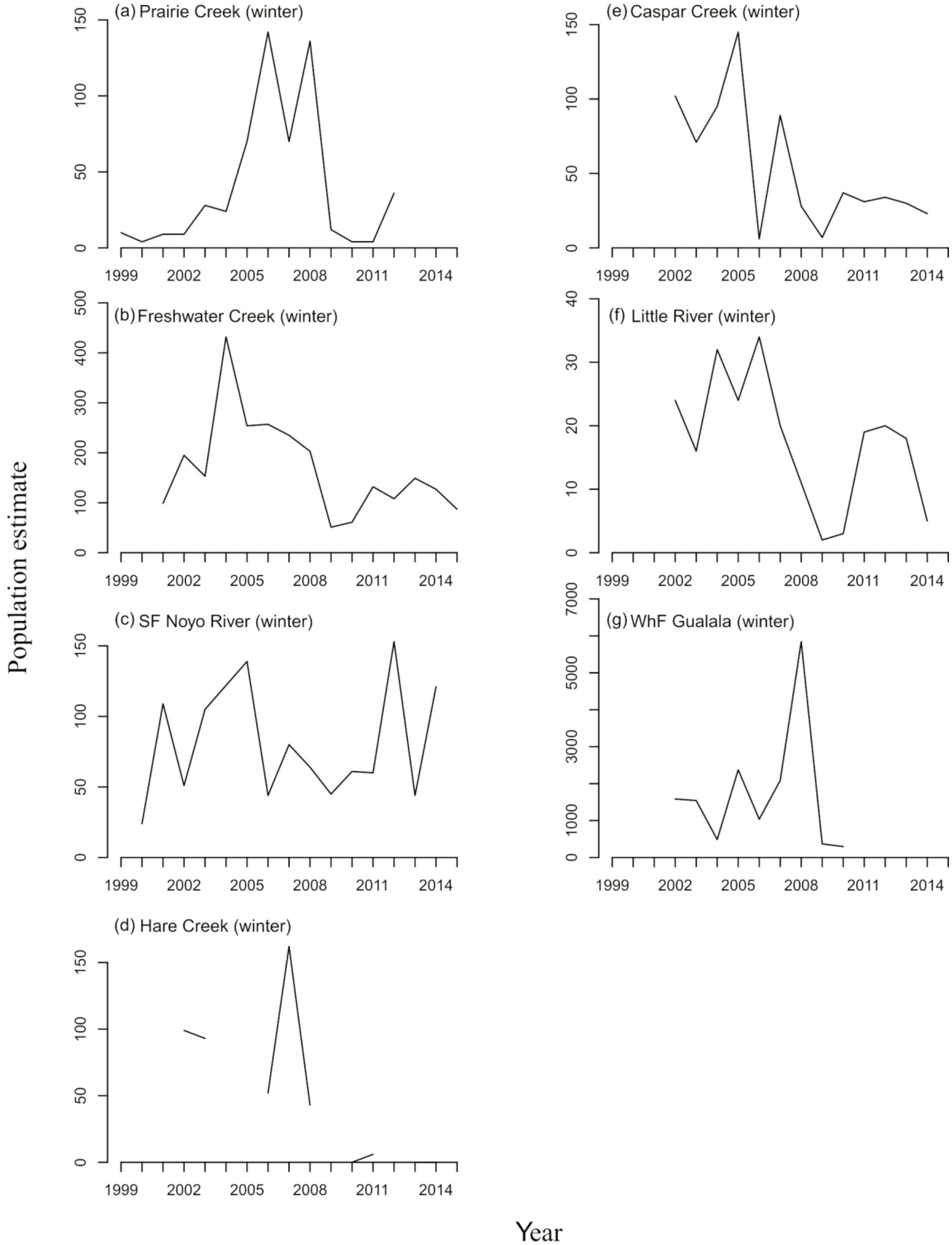

Figure 4.11. Time series of population abundance estimates for dependent populations or partial populations of winter-run NC-Steelhead. Estimates for Prairie Creek are based on the area-under-the-curve (AUC) method. Estimates for Wheatfield Fork Gualala River are based on counts of live fish observed from boat surveys. All other estimates are based on fish/redd expansions or mark-recapture estimates from life-cycle monitoring stations.

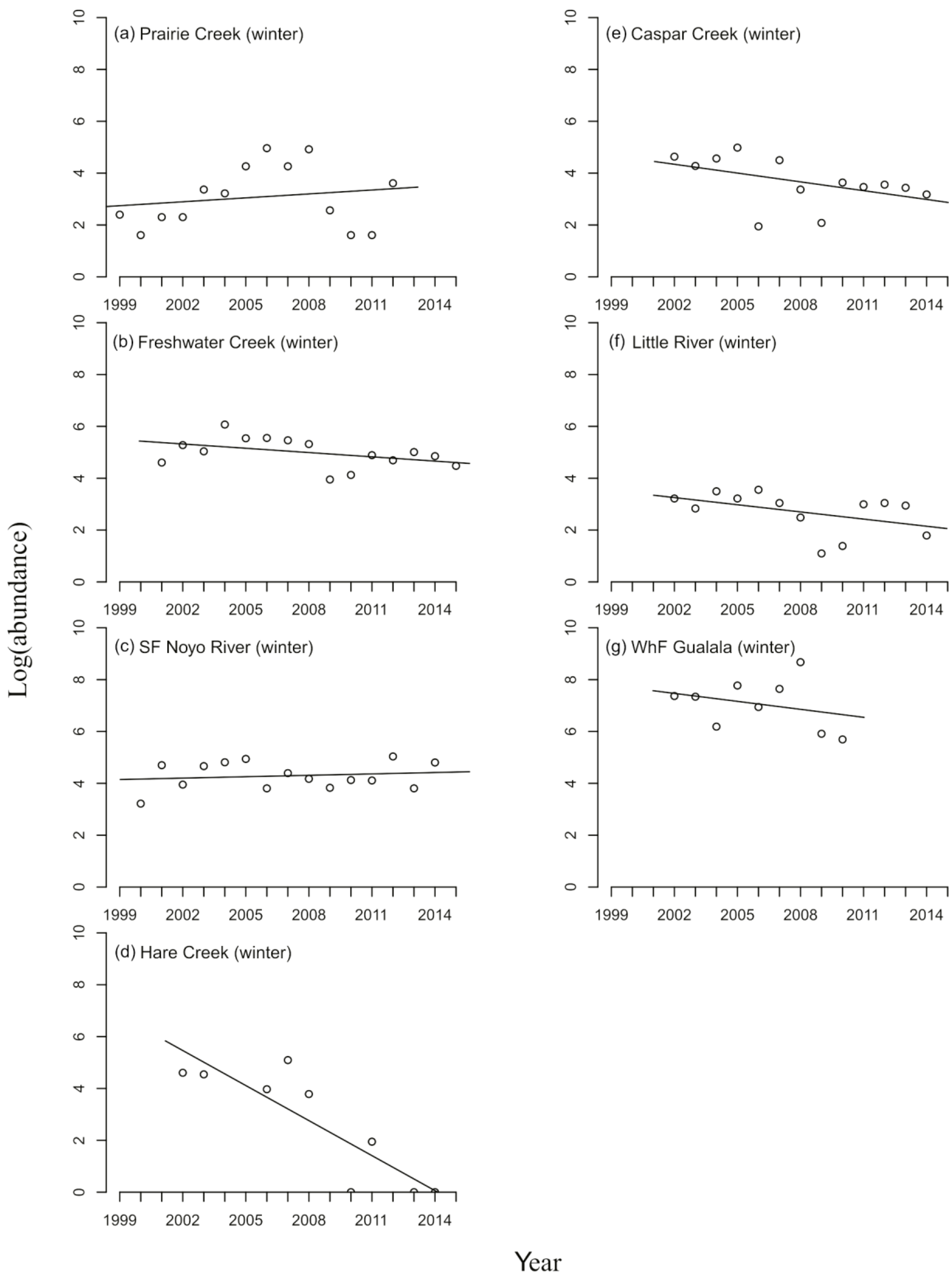

Figure 4.12. Population trends for dependent populations or partial populations of winter-run NC-Steelhead. Estimates for Prairie Creek are based on the area-under-the-curve (AUC) method. Estimates for Wheatfield Fork Gualala River are based on counts of live fish observed from boat surveys. All other estimates are based on fish/redd expansions or mark-recapture estimates from life-cycle monitoring stations.

Table 4.9. Population information for summer-run NC-Steelhead or populations with only index data or partial population estimates. NA indicates not available or applicable. Trends are shown only for populations where time series is at least six years, **bold** indicates significant trend.

Stratum/population	Years	$\overline{N}_{a(arith)}$	$\overline{N}_{a(geom)}$	$\overline{N}_{g(harm)}$	$\hat{T}$ (95% CI)	$\overline{N}_{g(harm)}$ target
Northern Coastal						
Redwood Creek[a]	34	10	7	18	-0.006 (-0.038, 0.027)	2500
	16	9	7	17	0.073 (-0.009, 0.154)	
Mad River	3	427	414	-	-	2500
SF Eel River	-	-	-	-	-	2500
Mattole River[b]	19	73	67	203	NA	2500
North Mtn. Interior						
Van Duzen River[c]	5	132	115	413	NA	2500
Larabee Creek	-	-	-	-	-	2500
NF Eel River	-	-	-	-	-	2500
Up-Mid Mainstem Eel River	-	-	-	-	-	2500
MF Eel River[d]	48	789	703	2107	-0.002 (-0.013, 0.008)	2500
	16	638	601	1428	0.049 (0.016, 0.081)	

a – The Redwood Creek summer steelhead population contributes to both the Northern Coastal and North Mountain Interior diversity strata. Estimates are from dive counts of a standardized reach and thus represent only a partial population estimate.
b – The Mattole River surveys cover only a portion of available rearing habitat and are thus a partial population estimate. Total stream miles surveyed is inconsistent from year to year; thus, calculation of trends was deemed inappropriate.
c – The Van Duzen River summer steelhead survey likely covers most of the available summer holding pools for the population.
d – The Middle Fork Eel River summer steelhead survey likely covers most of the available summer holding pools for the population.

Lower Interior Stratum. The Lower Interior Stratum includes eight populations of winter-run steelhead in tributaries that enter the Eel River primarily from the west and south between Jewett Creek and Soda Creek, inclusive. We are aware of no information on the status or viability of these populations (Table 4.7).

North Mountain Interior Stratum. The North Mountain Interior Stratum includes tributaries that enter the Eel River from the east from the Van Duzen River to the Middle Fork Eel River, and including the upper mainstem Eel River. The only dataset available for winter-run steelhead in this region are counts of steelhead at Van Arsdale Station, which represents a composite of the Bucknell Creek and Soda Creek populations (both considered part of the Lower Interior Stratum), as well as a small portion of the historical range of the Upper Mainstem Eel River population. Analysis of counts at Van Arsdale Station is confounded by a long history of hatchery activity within the basin and the inability to discriminate between hatchery-origin and natural-origin fish in the years preceding 1997. Over the 78-year period of record, an average of 1,854 steelhead have

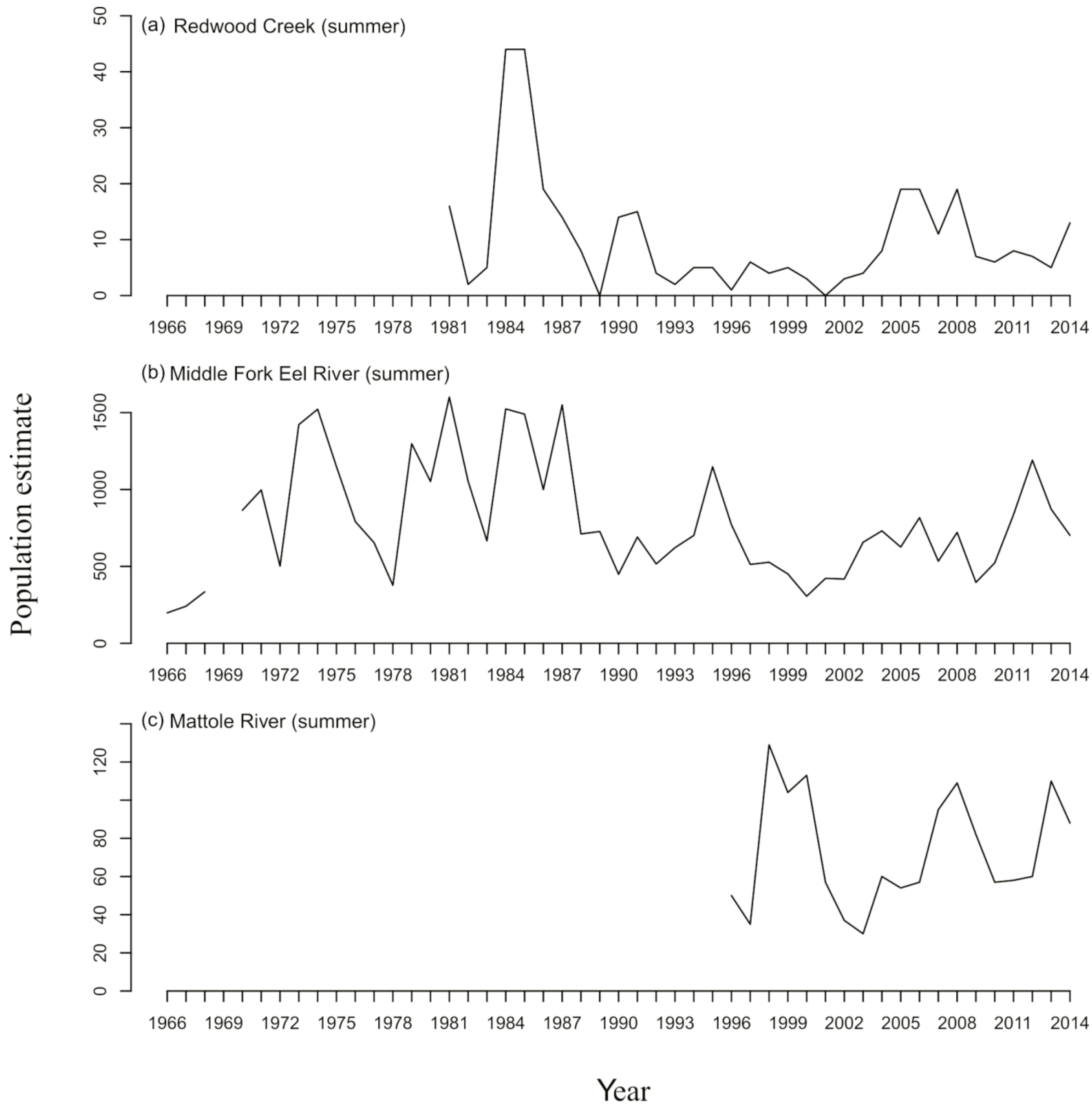

Figure 4.13. Time series of population abundance estimates for independent populations of summer-run NC-Steelhead. Estimates for Redwood Creek and Mattole River are summer dive counts for index reaches. Estimates from Middle Fork Eel River are based on summer dive counts covering most available oversummering habitat.

been counted at Van Arsdale (Table 4.8); however, the more recent (16-year) average has been 631 fish, with 328 of these being of natural origin. The long-term trend (combined natural-origin and hatchery-origin fish) has been negative ($p < 0.001$); however, the recent (16-year) trend for natural-origin fish has been positive ($p = 0.024$) (Table 4.8; Figures 4.15, 4.16). Without knowing which of the three populations these fish represent, it is difficult to evaluate these numbers against viability criteria for these populations.

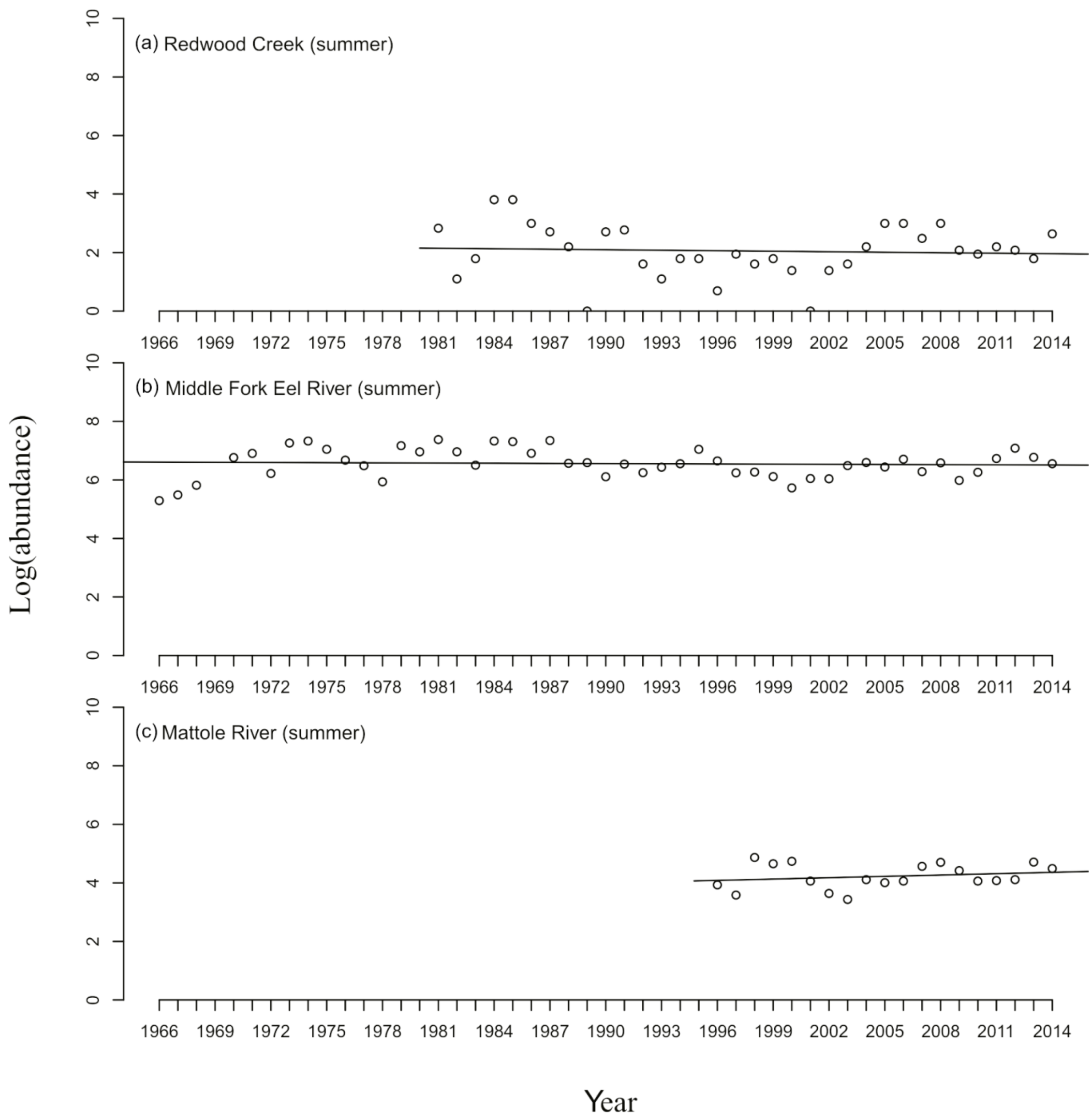

Figure 4.14. Population trends for independent populations of summer-run NC-Steelhead. Estimates for Redwood Creek and Mattole River are summer dive counts for index reaches. Estimates from Middle Fork Eel River are based on summer dive counts covering most available oversummering habitat.

Nevertheless, it is clear that neither Bucknell Creek nor the Upper Mainstem Eel River population is approaching viability targets. In the latter case, this is not surprising given that the majority of historical habitat lies above an impassable dam and the remaining habitat is insufficient to support a viable population.

For summer steelhead in this stratum, dive counts dating back to 1966 are available for the Middle Fork Eel River population. The long-term average abundance was 789 spawners with essentially no trend over the period of record ($p = 0.699$) (Table 4.9; Figures 4.13b, 4.14b). The recent (16-year) average has been slightly lower at 638 with a significant positive trend during that time ($p = 0.006$) (Table 4.9). Overall, the population is currently at about 60% of the viability target for this population. Recently, CDFW initiated summer dive surveys on the Van Duzen River. These surveys cover the reach between Little Larabee Creek and Eaton Roughs (generally considered the upper extent of anadromy on the mainstem Van Duzen River), which is thought to encompass the majority of available holding pools in the river[9] (S. Thompson, CDFW, personal communication). Over the past five years, an average of 132 (range 54–255) steelhead has been counted each year (Table 4.9). The population is currently at about 17% of the viability target for this population.

North-Central Coastal Stratum. The availability of information on steelhead abundance in the North-Central Coastal stratum has improved considerably since the CMP was fully implemented in 2009. Population estimates are now available for all nine independent populations in the stratum, though time series exceeding 6 years are available for only two of these populations (Pudding Creek and Noyo River). For most of the smaller watersheds, including Usal Creek, Cottaneva Creek, Wages Creek, Pudding Creek, and Big Salmon Creek, population estimates over the last 3–13 years have averaged between 60 and 100 fish, which ranges from 9% to 13% of the viability targets (Table 4.7). Of these five populations, trends were estimated only for those with 6 or more years of record. For Usal Creek, the trend was positive but not significant ($p = 0.186$) (Table 4.7; Figures 4.17a, 4.18a). For Pudding Creek, the trend over the last 13 years has been negative and significant ($p = 0.019$) (Table 4.7; Figures 4.17c, 4.18c). This trend is driven by four consecutive years (2009–2012) of returns of fewer than 30 spawners, which also accounts for the population falling below the high-risk depensation threshold ($D_{dep} = 0.7$) (Table 4.7).

For the four largest watersheds in the stratum, estimates of population abundance have been generally higher. In the Ten Mile River, estimates of steelhead adults have averaged 407 (range 0–869) over the last 6 years, with the short-term trend being positive and marginally significant ($p = 0.062$) (Table 4.7; Figures 4.17b, 4.18b). This population has fallen below the high-risk depensation threshold ($D_{dep} = 0.8$) but has since rebounded. Estimates of steelhead adults in the Noyo River over the last 13 years have averaged 343 fish (range 79–593), and have shown essentially no trend ($p = 0.435$) (Table 4.7; Figures

[9] Almost all observations of summer steelhead over the last 30 years have been made in this reach; however, in 2015, 29 adult summer steelhead were observed in the lower river near the mouth of Yager Creek. It is believed that this unusual occurrence was likely due to extreme drought conditions that limited upstream migration of summer steelhead this year (S. Thompson, CDFW, personal communication). Nevertheless, it raises the possibility that the dive counts may underestimate total population size in some years.

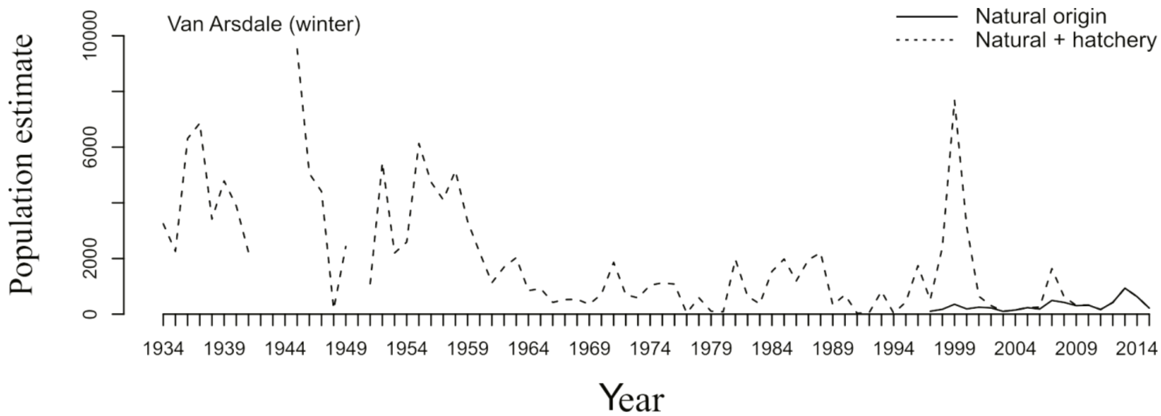

Figure 4.15. Time series of population abundance estimates for dependent populations or partial populations of winter-run NC-Steelhead. Estimate for Van Arsdale Station is a dam count potentially representing portions of multiple populations.

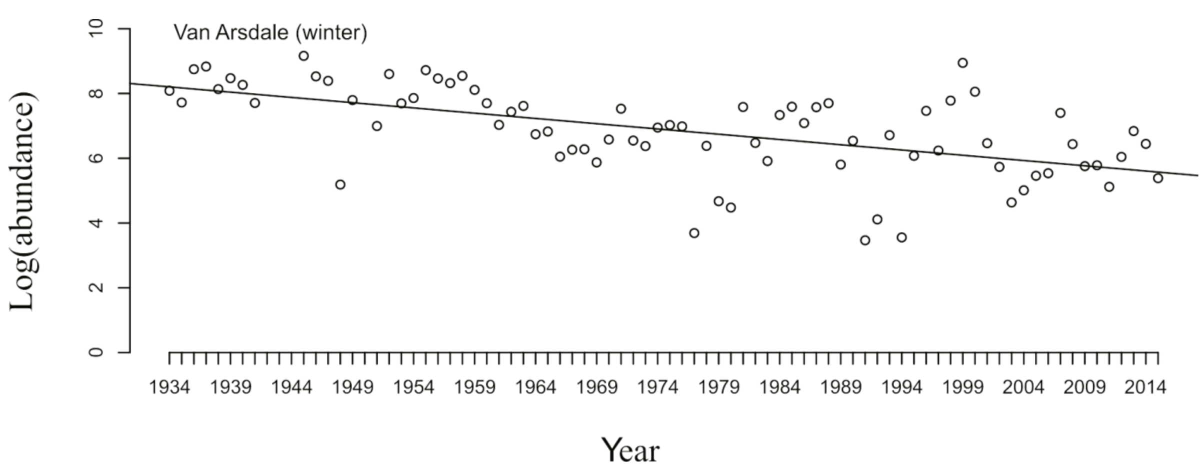

Figure 4.16. Population trends for dependent populations or partial populations of winter-run NC-Steelhead. Estimate for Van Arsdale Station is a dam count potentially representing portions of multiple populations.

4.17d, 4.18d;). Estimates for Big River have averaged 633 (range 52–1,820) over the past 6 years. The population trend has been positive ($p = 0.002$) (Table 4.7; Figures 4.17e, 4.18e); however, the population also falls below the high-risk depensation threshold ($D_{dep} = 0.4$) as a consequence of low abundance in the first three years of the time series (Table 4.7). Finally, the Albion River has averaged 60 adults (range 13–182) over the last 6 years. This population has also shown a positive short-term trend ($p = 0.043$) but has also dipped below the high-risk depensation threshold ($D_{dep} = 0.3$) (Table 4.7; Figures 4.17f, 4.18f;). For all four of these populations, the estimated abundances lie between 4% and 12% of viability targets.

Data are also available for four dependent or partial populations in this stratum. Population estimates for three dependent populations, Hare Creek, Caspar Creek, and Little River, over the last 9–13 years have averaged between 18 and 54 fish (Table 4.8), and trends for all three have been negative, though significantly so only for Hare Creek ($p = 0.003$) (Table 4.8; Figures 4.11d-f, 4.12d-f). The estimate from the South Fork Noyo River (part of the Noyo River population) has averaged 81 adults (range 24–153), with essentially no trend over the 15 years of record ($p = 0.585$) (Table 4.8; Figures 4.11c, 4.12c).

Central Coastal Stratum. Population estimates are now available for four of five independent populations in the Central Coastal Stratum, though in all cases, the time series span 6 or fewer years. The estimated return of steelhead adults to the Navarro River has averaged 366 (range 102–781) over 6 years (Table 4.7). The trend over that time has been positive and significant ($p = 0.017$); however, the population remains at only 5% of the viability target and fell below the high-risk depensation threshold in the early part of the time series (Table 4.7; Figures 4.17g, 4.18g). Elk Creek has been sampled only two of the past 6 years, producing an average of 31 adult steelhead during those years (range 3–59) (Table 4.7). Brush Creek has produced an average of 13 steelhead adults in the past 6 years (range 0–41), with a positive but nonsignificant ($p = 0.305$) trend (Table 4.7, Figures 4.17h, 4.18h). This population is also well below the high-risk depensation threshold ($D_{dep}=0.1$). Finally, the Garcia River has produced an estimated 326 steelhead adults annually (range 65–773) for the past 6 years and also shows a positive but nonsignificant trend ($p = 0.366$) (Table 4.7; Figures 4.17i, 4.18i).

Population estimates for only two years are available for the North Fork Navarro River (part of the Navarro River population) and Greenwood Creek. The North Fork Navarro River has produced an estimated 358 spawners annually (range 251–466), while Greenwood Creek has produced an average of 7 spawners (range 0–14) (Table 4.8). Outside of the CMP effort, estimates of adult steelhead in the Wheatfield Fork of the Gualala River based on direct observation of adults in holding pools were generated from 2002 to 2010 (DeHaven 2010). These efforts produced estimates averaging 1,735 adults annually (range 296–5,843) (Table 4.8). These data indicate a negative but nonsignificant ($p = 0.452$) trend. Though only a partial population estimate, these data suggest that the Gualala River population is perhaps the largest remaining in the Central Coastal Stratum and perhaps the DPS as a whole. Regrettably, this monitoring effort was discontinued after 2010 and there is no new information on this population.

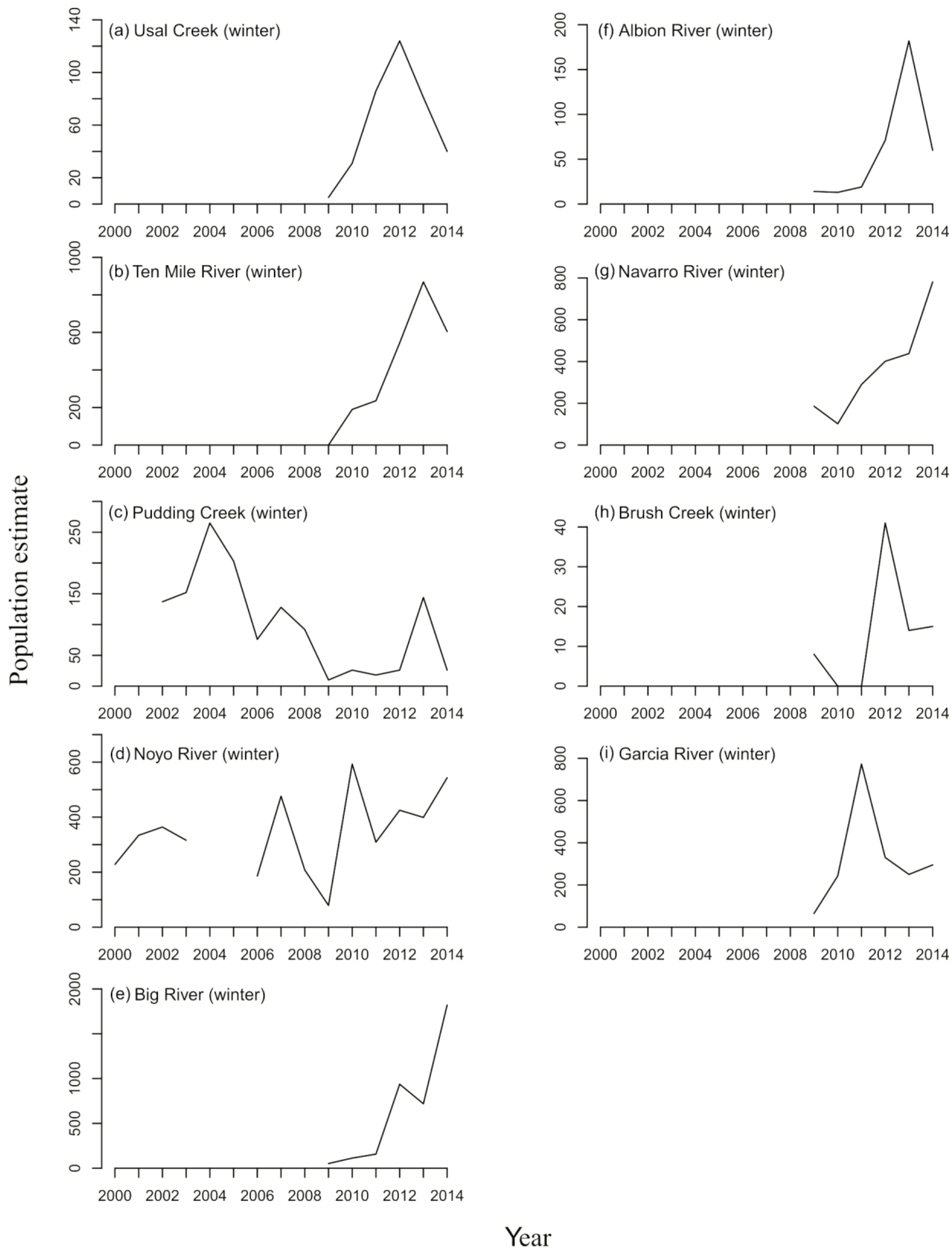

Figure 4.17. Time series of population abundance estimates for independent populations of winter-run NC-Steelhead. All estimates are based on fish/redd expansions from life-cycle monitoring stations.

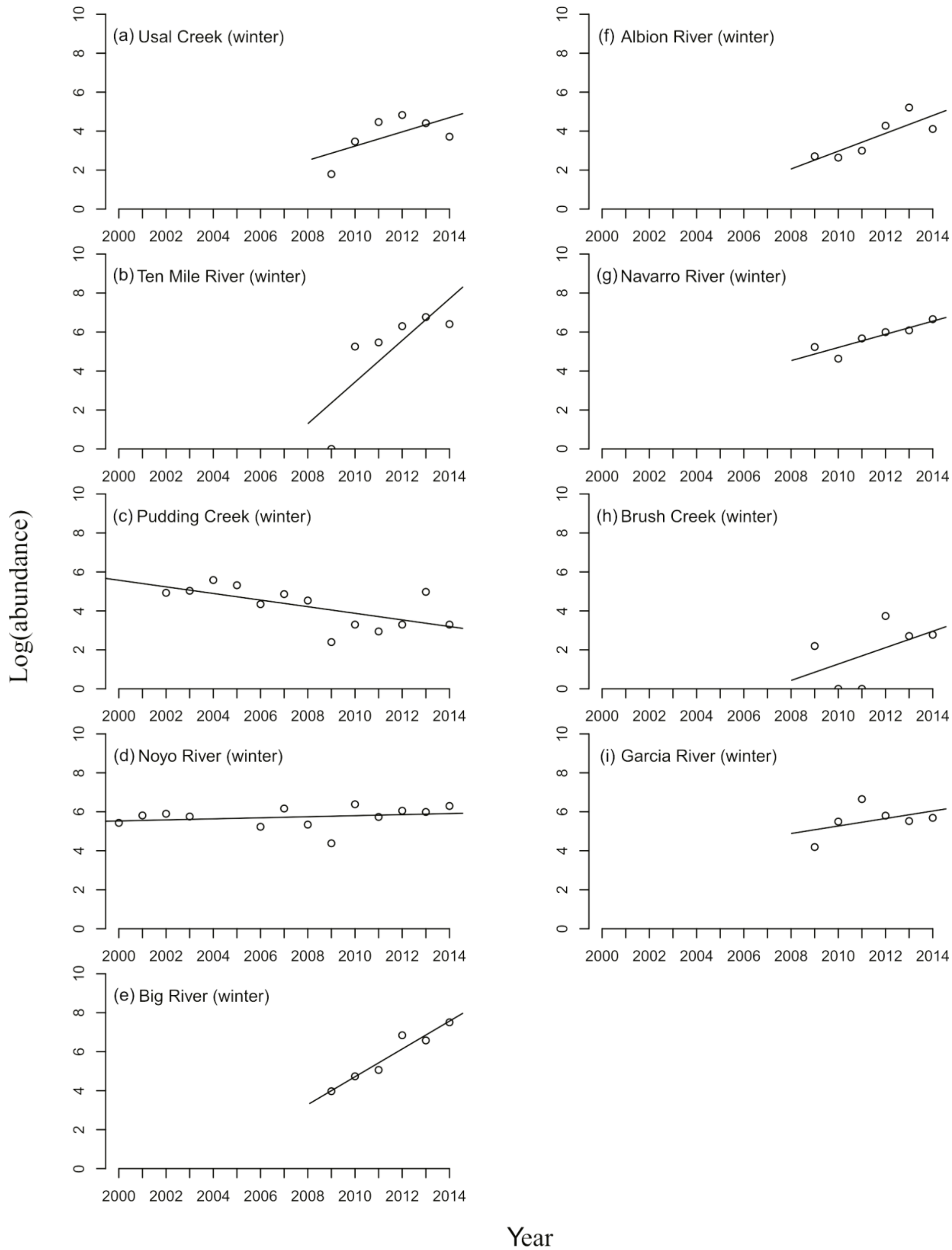

Figure 4.18. Population trends (log abundance) for independent populations of winter-run NC-Steelhead. All estimates are based on fish/redd expansions from life-cycle monitoring stations.

Harvest Impacts[10]

Ocean harvest of steelhead is extremely rare, and is in particular an insignificant source of mortality for NC-steelhead. While insufficient data exist to estimate NC-steelhead freshwater exploitation rates, these rates are likely relatively low given that retention of natural-origin steelhead is prohibited in California. Fishing effort estimates based on angler self-report cards are available for 2000–2014 (Figure 4.19). Beginning in 2013, fishing regulations for many streams changed from allowing no steelhead retention to allowing a daily bag limit of two hatchery-origin steelhead per day. In summary, while no direct information is available on the level of NC steelhead fishery impacts, it is reasonable to conclude that the level of impact has either not appreciably changed since the 2010 salmon and steelhead assessment (Williams et al. 2011), or potentially increased due to increased bag limits for hatchery-origin fish.

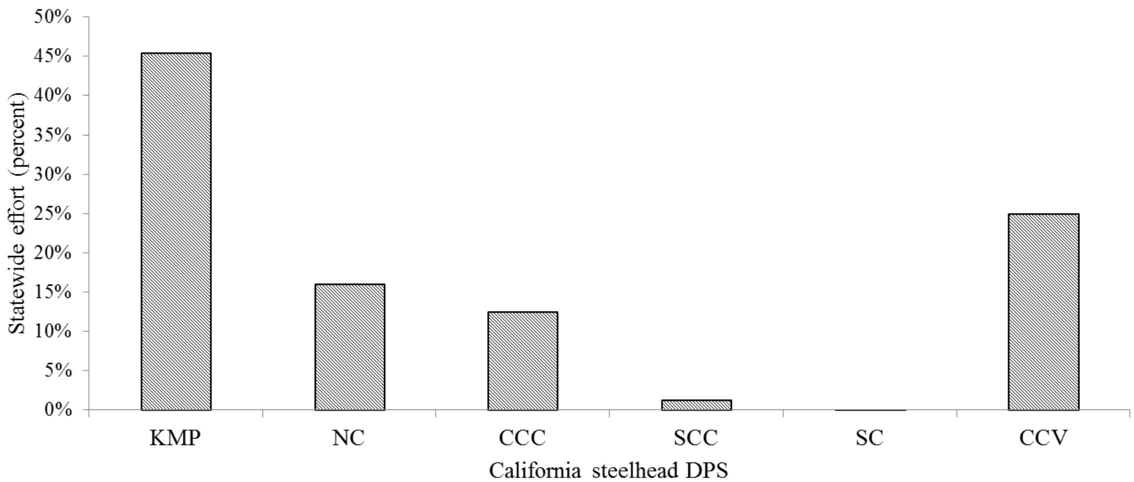

Figure 4.19. Distribution of California statewide steelhead fishing effort by DPS for years 2000−2014 (Jackson 2007; Farhat in preparation).

Summary and Conclusions

The availability of information on steelhead populations in the NC-Steelhead DPS has improved considerably in the past 5 years, thanks to implementation of the CMP across a significant portion of the DPS. Nevertheless, significant gaps in information still remain, particularly in the Lower Interior and North Mountain Interior diversity strata, where there is very little information from which to assess viability. Overall, the available data for winter-run populations—predominately in the North Coastal, North-Central Coastal, and Central Coastal strata—indicate that all populations are well below viability targets, with most being between 5% and 13% of these goals. For the two Mendocino Coast

[10] Harvest impacts section prepared by Michael O'Farrell

populations with the longest time series, Pudding Creek and the Noyo River, the 13-year trends have been negative and neutral, respectively (Figures 4.18c, 4.18d). However, the short-term (6-year) trend has been generally positive for all independent populations in the North-Central Coastal and Central Coastal strata, including the Noyo River and Pudding Creek (Figure 4.18). Data from Van Arsdale Station likewise suggests that, although the long-term trend has been negative, run sizes of natural-origin steelhead have stabilized or are increasing. Thus, we have no strong evidence to indicate conditions for winter-run have worsened appreciably since the last assessment (Williams et al. 2011).

Summer-run populations continue to be of significant concern. The Middle Fork Eel River population has remained remarkably stable for nearly five decades and is closer to its viability target than any other population in the DPS (Table 4.9). Although the time series are short, the Van Duzen River and Mad River appear to be supporting populations numbering in the low hundreds. However, the Redwood Creek and Mattole River populations appear small, and little is known about other populations including the various tributaries of the Eel River (i.e., Larabee Creek, North Fork Eel River, and South Fork Eel River).

In summary, the available information for winter-run and summer-run populations of NC Steelhead do not suggest an appreciable increase or decrease in extinction risk since publication of the last viability assessment (Williams et al. 2011). Most populations for which there are population estimates available remain well below viability targets; however, the short-term increases observed for many populations, despite the occurrence of a prolonged drought in northern California, suggests this DPS is not at immediate risk of extinction.

4.4 Central California Coast Steelhead

DPS Boundary Delineation

See discussion of steelhead DPS boundary issues in introduction.

Summary of Previous Assessments

The original BRT concluded that the Central California Coast (CCC) Steelhead DPS was in danger of extinction (Busby et al. 1996), citing extreme risk for populations in Santa Cruz County and tributaries to San Francisco and San Pablo bays, as well as apparent substantial declines in numbers and threats to genetic integrity (caused by hatchery activities) in the Russian River. A subsequent status review (NMFS 1997) concluded that the ESU was not presently in danger of extinction but was likely to become so in the foreseeable future; the change in opinion of the BRT was prompted by new data showing that steelhead remained present in most watersheds in the Santa Cruz Mountains and were more abundant than previously thought. This DPS was listed as threatened in late 2007 (62 FR 43937). Good et al. (2005) similarly concluded that the DPS was not presently in danger of extinction, but was likely to become so in the foreseeable future, and the DPS's status as threatened was reaffirmed (71 FR 834). The general paucity of data was identified as a continuing source of uncertainty in these reviews. In the most recent assessment, Williams et al. (2011) concluded that there was little information available to indicate a change in the viability of this DPS, though again acknowledged the high uncertainty surrounding most populations, particularly those entering San Francisco and San Pablo bays.

New Data and Updated Analyses

Steelhead populations in the CCC-Steelhead DPS are the most poorly monitored salmonid populations in the NCCC Recovery Domain. Population-level estimates of adult abundance are entirely lacking for 28 populations that constitute the North Coastal, Interior, Coastal San Francisco Bay, and Interior San Francisco Bay diversity strata. Only in the Santa Cruz Mountain stratum has implementation of the CMP been initiated, and here only recently. Thus, with the exception of the life-cycle monitoring station in Scott Creek, estimates of abundance span only 1–3 years for populations in this stratum. More limited monitoring efforts have produced data for a few partial populations, but the lack of data continues to make it extraordinarily difficult to assess the status, trends, and viability of populations in the DPS. We summarize the limited information below by stratum.

North Coastal Stratum. This stratum includes tributaries in the lower Russian River watershed downstream of the confluence of Mark West Creek, as well as coastal watersheds of Sonoma and Marin counties. There are no comprehensive efforts to monitor any of the independent or dependent populations in this stratum. Spawner surveys have been conducted in the Lagunitas Creek watershed since 2001-2002; however, these target coho salmon and do not encompass the full spawning period of steelhead. Consequently, the redd counts are not considered reliable indicators of trends. With those caveats in mind, redd counts for this period, which perhaps serve as a minimum estimate for spawners, have averaged approximately 155 (range 23–320) (Ettlinger et al. 2015). Given the incomplete nature of these surveys and lack of developed methods for expanding redd counts to adult estimates, it is difficult to compare these values with viability targets. However, these redd counts suggest that the population is well below its viability target of 1900 adults (Table 4.10).

Redd surveys for two dependent populations in this stratum, Redwood Creek and Pine Gulch, are conducted by the National Park Service. As with the Lagunitas Creek surveys, these surveys target coho salmon and thus do not encompass the full spawning period for steelhead. A rough estimate of returning adults has been made by multiplying the redd count by two (assumes one redd per female, and one male per female). Over 14 years, the average estimate has been 17 for Pine Gulch and 13 for Redwood Creek (Table 4.11). Trends for both of these time series have been positive but nonsignificant (Table 4.11; Figures 4.20a-b, 4.21a-b).

Interior Stratum. The Interior Stratum of this DPS consists of populations in the upper Russian River basin, upstream and inclusive of Mark West Creek. We know of no systematically collected data on naturally produced steelhead adults for this stratum (Table 4.10). Warm Springs Hatchery and Coyote Valley Fish Facility continue to produce approximately 660,000 yearling steelhead annually as part of mitigation for the loss of steelhead habitat behind Warm Springs and Coyote dams (Clifford 2015, unpublished data), and these fish are distributed throughout the upper and lower watershed. In the last 15 years, an average of approximately 6,300 steelhead have returned to the hatchery annually (Coey 2015), the majority of these (> 95%) being marked fish of hatchery origin. The lack of spawner surveys on natural spawning grounds within the upper Russian River basin make it impossible to assess either the abundance of natural-origin fish or the fraction of fish on spawning grounds that are of hatchery origin.

Coastal San Francisco Bay Stratum. Population-level estimates of adult abundance are not available for any of the seven independent populations within this stratum. Nor is there any population information for dependent populations within this stratum. Adult steelhead are periodically reported in several creeks, including San Francisquito Creek (M. Stoecker, Stoecker Ecological, personal communication) and Miller Creek (Marin County Watershed Program 2015). However, information is insufficient to evaluate whether there has been any change in viability.

Interior San Francisco Bay Stratum. Population-level estimates of adult abundance are also lacking for all 10 independent populations of steelhead in the Interior San Francisco Bay Stratum. Spawner surveys have been conducted in recent years in selected portions of the Napa River watershed and have produced occasional sightings of steelhead redds,

Table 4.10. Viability metrics for independent populations of steelhead in the CCC-Steelhead DPS. NA indicates not available or applicable. Trends shown only for populations where time series is at least six years, **bold** indicates significant trend. IPkm includes only habitats that are currently accessible. $N_{a(arith)}$ target refers to the low-risk viability target identified by the Technical Recovery Team (Spence et al. 2008).

Stratum/population	Years	$\overline{N}_{a(arith)}$	$\overline{N}_{a(geom)}$	$\overline{N}_{g(harm)}$	$\hat{C}$	$\hat{T}$ (95% CI)	IPkm	$\hat{D}_{dep}$	$\hat{D}_{ssd}$	$\hat{D}_{ssd}$ target	$\overline{N}_{a(arith)}$ target
North Coastal											
Austin Creek	-	-	-	-	-	-	95.4	-	-	29.0	2800
Green Valley Creek	-	-	-	-	-	-	37.0	-	-	37.1	1400
Salmon Creek	-	-	-	-	-	-	36.6	-	-	37.1	1400
Americano Creek	-	-	-	-	-	-	35.4	-	-	37.3	1300
Stemple Creek	-	-	-	-	-	-	45.1	-	-	36.0	1600
Walker Creek	-	-	-	-	-	-	57.8	-	-	34.2	2000
Lagunitas Creek							53.8			34.7	1900
Interior											
Mark West Creek	-	-	-	-	-	-	271.9	-	-	20.0	5400
Dry Creek	-	-	-	-	-	-	116.4	-	-	20.0	3000
Maacama Creek	-	-	-	-	-	-	76.1	-	-	31.6	2400
Upper Russian River	-	-	-	-	-	-	542.4	-	-	20.0	10800
Santa Cruz Mtns											
Pilarcitos Creek	-	-	-	-	-	-	20.7	-	-	39.4	800
San Gregorio Creek[a]	2	136	135	NA	NA	-	55.2	-	2.5	34.6	1900
Pescadero Creek[a]	3	591	361	1773	NA	-	66.4	8.9	8.9	33.0	2200
Waddell Creek[a]	2	74	73	NA	NA	-	13.7	-	5.4	40.0	500
Scott Creek[b]	12	202	174	518	0.55	**-0.136 (-0.197, -0.075)**	18.9	5.4	10.7	39.6	700
Laguna Creek	-	-	-	-	-	-	13.1	-	-	40.0	500
San Lorenzo River[a]	3	525	423	1575	NA	-	153.0	3.4	3.4	21.0	3200

Table 4.10. continued.

Stratum/population	Years	$\overline{N}_{a(arith)}$	$\overline{N}_{a(geom)}$	$\overline{N}_{g(harm)}$	$\hat{C}$	$\hat{T}$ (95% CI)	IPkm	$\hat{D}_{dep}$	$\hat{D}_{ssd}$	$\hat{D}_{ssd}$ target	$\overline{N}_{a(arith)}$ target
Soquel Creek[a]	1	8	8	NA	NA	-	54.2	-	0.1	34.7	1900
Aptos Creek[a]	1	70	70	NA	NA	-	29.7	-	2.4	38.1	1100
Coastal SF Bay											
Corte Madera Creek	-	-	-	-	-	-	26.4	-	-	38.6	1000
Novato Creek	-	-	-	-	-	-	39.1	-	-	36.8	1400
Guadalupe River	-	-	-	-	-	-	87.2	-	-	30.1	2600
Saratoga Creek	-	-	-	-	-	-	2.4	-	-	-	-
Stevens Creek	-	-	-	-	-	-	14.5	-	-	-	-
San Francisquito Creek	-	-	-	-	-	-	28.8	-	-	38.2	1100
San Mateo Creek	-	-	-	-	-	-	7.7	-	-	-	-
Interior SF Bay											
Petaluma River	-	-	-	-	-	-	147.7	-	-	21.7	3200
Sonoma Creek	-	-	-	-	-	-	198.1	-	-	20.0	4000
Napa River	-	-	-	-	-	-	357.0	-	-	20.0	7100
Green Valley/Suisun Cr	-	-	-	-	-	-	82.4	-	-	30.8	2500
Walnut Creek	-	-	-	-	-	-	5.6	-	-	-	-
San Pablo Creek	-	-	-	-	-	-	10.1	-	-	-	-
San Leandro Creek	-	-	-	-	-	-	11.9	-	-	-	-
San Lorenzo Creek	-	-	-	-	-	-	24.6	-	-	38.8	1000
Alameda Creek	-	-	-	-	-	-	24.8	-	-	38.8	1000
Coyote Creek	-	-	-	-	-	-	140.5	-	-	22.7	3200

a – Numbers indicate the estimated number of adults based on fish/redd expansions from life-cycle monitoring stations.
b – Mark-recapture estimates from Scott Creek life-cycle monitoring station.

Table 4.11. Viability metrics for dependent populations of steelhead in the CCC-Steelhead DPS. NA indicates not available or applicable. Trends shown only for populations where time series is at least six years, **bold** indicates significant trend. $N_{a(arith)}$ targets have not been defined yet.

Stratum/population	Years	$\overline{N}_{a(arith)}$	$\overline{N}_{a(geom)}$	$\overline{N}_{g(harm)}$	$\hat{T}$ *(95% CI)*	$\overline{N}_{a(arith)}$ target
North Coastal						
Pine Gulch[a]	14	17	8	29	0.131 (-0.070, 0.332)	-
Redwood Creek[a]	19	13	6	-	0.188 (0.102, 0.274)	-
Santa Cruz Mtn						
San Pedro Creek[b]	1	38	38	NA	-	-
Gazos Creek[b]	3	58	30	175	-	-
San Vicente Creek[b]	3	61	35	182	-	-

a – Estimates are redd counts multiplied by 2.
b – Numbers indicate the estimated number of adults based on fish/redd expansions from life-cycle monitoring stations.

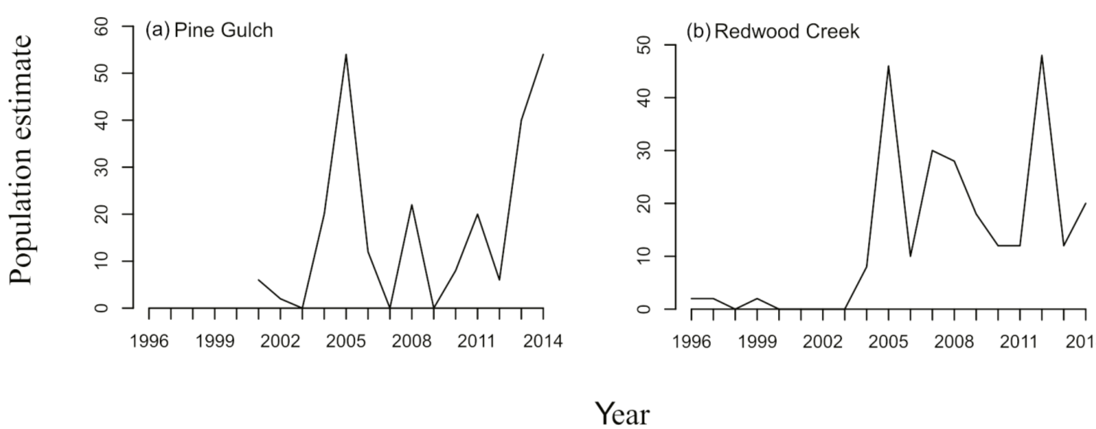

Figure 4.20. Time series of population abundance estimates for dependent populations of winter-run CCC-Steelhead. Estimates are two times the total redd count for the watershed.

live fish, or carcasses in the mainstem Napa Creek as well as three tributaries: York Creek, Heath Creek, and Redwood Creek (Koehler 2008; Koehler and Blank 2013). Additionally, a rotary screw trap operated near the upper limit of tidal influence has resulted in capture of 31 to 251 smolts annually since 2009 (Koehler 2014). These efforts confirm the occurrence of steelhead in this watershed. However, the highly limited spatial

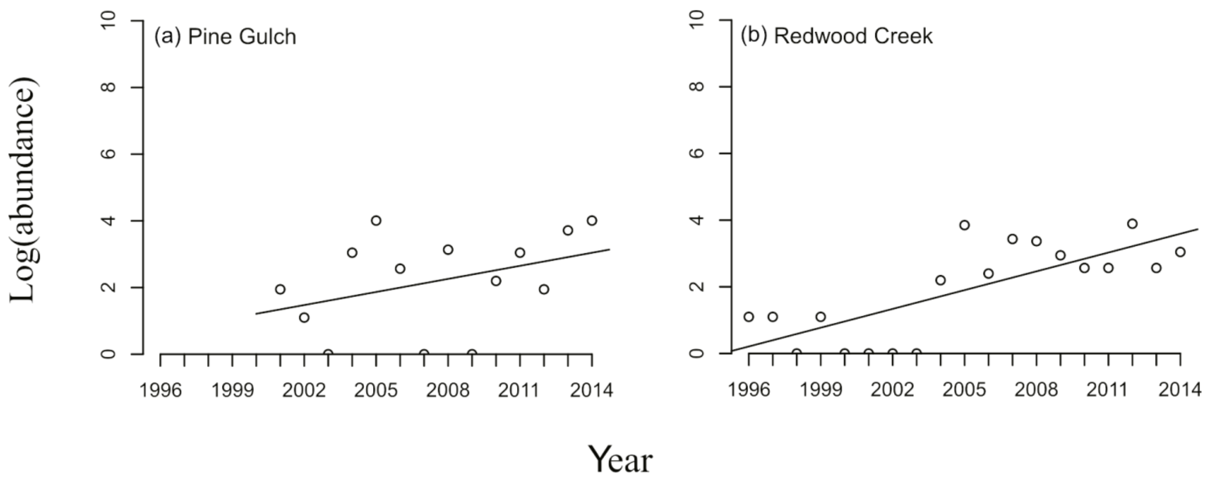

Figure 4.21. Population trends (log abundance) for dependent populations of winter-run CCC-Steelhead. Estimates are based on two times the total redd count for the watershed.

and temporal extent of the adult surveys and the lack of mark-recapture estimates that would allow expansion of smolt counts to population estimates do not allow any conclusions to be drawn about population status or trends. Likewise, limited spawner surveys in selected tributaries of the Petaluma River produced 6 live steelhead, 2 carcasses, and 6 redds, all in Adobe Creek during the 2013–2014 spawning season (Morrison et al. 2014). Again these limited surveys confirm steelhead presence in the watershed, but do not allow conclusions to be drawn about current viability.

Santa Cruz Mountains Stratum. The Scott Creek LCM station provides the only population estimates of adult steelhead abundance in the entire CCC Steelhead DPS for a period spanning more than 3 years. Over the past twelve years, an average of 202 steelhead adults have returned to this watershed, which is approaching 30% of the viability target. However, the population trend has been negative ($p < 0.001$) (Table 4.10; Figures 4.22, 4.23). Implementation of the coastal monitoring plan has produced estimates of steelhead in several other watersheds in this stratum, but only for the past 1–3 years. Results from these surveys indicate that populations in the three largest watersheds number in the hundreds of fish, from 136 in San Gregorio Creek to more than 500 in Pescadero Creek and the San Lorenzo River (Table 4.10). These values range from 7% (San Gregorio) to 27% (Pescadero) of the viability targets for these populations. Estimates for the smaller watersheds range from 8 fish in Soquel Creek (based on a single year of data) to over 70 fish in Waddell and Aptos creeks (Table 4.10). These values range from <1% to 15% of the populations’ viability targets.

Harvest Impacts[11]

Ocean harvest of steelhead is extremely rare, and is in particular an insignificant source of mortality for CCC-steelhead. While insufficient data exists to estimate CCC-Steelhead freshwater exploitation rates, these rates are likely relatively low given California's retention prohibition of natural-origin steelhead. Fishing effort estimates based on angler self-report cards are available for 1993–2005, which suggest that effort declined in the second half of the period in this DPS (Figure 4.19). Fishing effort estimates for more recent years are not available but there has been little change in the fishing opportunity status quo. However, beginning in 2013, fishing regulations for many streams changed from allowing no steelhead retention to allowing a daily bag limit of two hatchery-origin steelhead per day. Additionally, recent drought conditions have affected some steelhead fishing opportunities for this DPS. For example, the California Fish and Game Commission imposed an emergency fishery closure on the Russian River in February of 2014. The closure ended in April of that year. In summary, while no direct information is available on the level of CCC-Steelhead fishery impacts, it is reasonable to conclude that the level of impact has either not appreciably changed since the 2010 salmon and steelhead viability assessment (Williams et al. 2011), or potentially increased due to increased bag limits for hatchery-origin fish.

Summary and Conclusions

The scarcity of information on steelhead abundance in the CCC-Steelhead DPS continues to make it difficult to assess whether conditions have changed appreciably since the previous assessment of Williams et al. (2011), which concluded that the population was likely to become endangered in the foreseeable future. In the North Coastal and Interior strata, steelhead still appear to occur in the majority of watersheds, though in the Russian River basin, the ratio of hatchery fish to natural-origin fish returning to spawn remains largely unknown and continues to be a source of concern. New information from three years implementation of the CMP in the Santa Cruz Mountains Diversity Stratum indicates that population sizes are perhaps higher than previously thought. However, the downward trend in the Scott Creek population, which has the most robust estimates of abundance, is a source of concern. The viability of populations in the two San Francisco Bay diversity strata remains highly uncertain, and it is likely that many populations where historical habitat is now inaccessible due to dams and other passage barriers are likely at high risk of extinction.

In summary, while data availability for this DPS remains poor, we find little new evidence to suggest that the extinction risk for this DPS has changed appreciably in either direction since publication of the last viability assessment (Williams et al. 2011).

[11] Harvest impacts section prepared by Michael O'Farrell

5 Central Valley Recovery Domain

Rachel C. Johnson and Steven T. Lindley

Several important planning efforts have been completed since the last viability assessment, including a Federal recovery plan for Sacramento River Winter-run Chinook salmon (SRWRC), Central Valley Spring-run Chinook salmon (CVSRC), and Central Valley Steelhead (NMFS 2014b). The recovery plan draws on the expertise of the Central Valley Technical Recovery Team (TRT) and is guided by the scientific framework and foundation provided by Lindley et al. (2006, 2007). The recovery plan along with the science and restoration actions identified in the biological opinion for the long-term operations of the Central Valley Project and State Water Project are key decision-making documents for improving and sustaining the health of California's salmon resources (NMFS 2009a, NMFS 2014b). The further development and implementation of life-cycle models for SRWRC and CVSRC will be seminal advancements in our understanding of how water project operations and restoration actions outlined in the recovery plans influence salmon population dynamics and long-term population viability (Hendrix et al. 2014).

5.1 Sacramento River Winter-run Chinook Salmon ESU

ESU Boundary Delineation

The Sacramento River Winter-run Chinook Salmon (SRWRC) ESU includes winter-run Chinook salmon spawning in the mainstem Sacramento River below Keswick Dam and Livingston Stone National Fish Hatchery (LSNFH). No new information suggests that the boundary of this ESU should change or that its status as an ESU should change.

Summary of Previous Assessments

Good et al. (2005) concluded that the status of SRWRC ESU was endangered. The major concerns of the Biological Review Team (BRT) were that there is only one extant population, and it is outside of its historical spawning distribution in an artificially maintained habitat that is vulnerable to drought. In the most recent assessment, Williams et al. (2011) found that the viability of the ESU had changed little since the 2005 review and found that it did not appear that there was a change in extinction risk.

Brief Review of TRT Documents and Previous Findings

The TRT delineated four historical independent populations of SRWRC. The spawning areas of three of these historical populations are above the impassable Keswick and Shasta dams, while Battle Creek (location of the fourth population) is presently unsuitable for winter-run Chinook salmon due to high summer water temperatures. Lindley et al. (2007) developed viability criteria for Central Valley salmonids, summarized in Table 5.1. Using data through 2004, Lindley et al. (2007) found that the mainstem Sacramento River population was at low risk of extinction. The ESU as a whole, however, could not be considered viable because there is only one naturally spawning population, and it is not spawning within the range of its historical spawning habitat. An emerging concern was rising levels of LSNFH-origin fish spawning in natural areas (mean=8%; t=10 years). However, the duration and extent of this introgression was still consistent with a low extinction risk as of 2010.

New Data and Updated Analyses

Since the 2010 viability assessment, routine escapement data have continued to be collected allowing viability statistics to be updated (Table 5.2). The Red Bluff Diversion Dam (RBDD) gates were operated in the up/out position during some or all of the winter-run immigration period since 2001, but were since removed in 2012 to provide unimpaired salmon passage year-round which changed the ability to count SRWRC adults at the RBDD fish ladders (NMFS 2009a). Population estimates from 2001 to present are derived exclusively from mark-recapture estimates from carcass surveys (Figure 5.1).

Table 5.2 shows the viability metrics for SRWRC abundance and trends in the LSNFH and in the Sacramento River. Like many other populations of Chinook salmon in the Central Valley, SRWRC have declined in abundance since 2005 with recent decadal lows of 827 spawners in 2011 (Figure 5.1). Escapement in 2011 represents the lowest run size since the construction and operation of the LSNFH in 1997. Both the current total population size (N; LSNFH = 645; Sacramento River = 11,125) and mean population sizes ($\hat{S}$; LSNFH = 215; Sacramento River = 3,708) satisfy the low risk criterion ($N > 2500$).

However, the point estimate for the 10-year trend in run size is negative (-0.15), suggesting a 15% per year decline in the population (Table 5.2). The slope is marginally not different than ‘0’, yet it is clear that the population has been steadily declining rather than increasing over the past decade. The maximum year-to-year decline in population size has reached 67%, an increase from 38% in the previous 2010 viability assessment (Williams et al. 2011). However, the percent decline does not exceed the catastrophic decline criteria (>90% decline in one generation nor annual run size < 500 spawners; Lindley et al. 2007).

These observed levels of hatchery influence exceed the low-extinction risk criteria met in the previous viability assessment and place the genetic integrity of the population at a moderate risk of extinction (Lindley et al. 2007). Since the beginning of hatchery production at LSNFH in 1997, the proportion of hatchery-origin SRWRC spawning in the river has increased (Figure 5.1). Prior to 2005, the proportion of LSNFH-origin spawners in the river was between 5% to 10%, consistent with guidelines from the Hatchery Scientific Review Group for conservation hatcheries (Figure 5.2; California HSRG 2012). However, the hatchery proportion has increased since 2005 and reached ~20% in 2005, 2014 and >30% in 2012. The average over the last 12 years

Table 5.1. Criteria for assessing the level of extinction risk for populations of Pacific salmonids in the Central Valley of California. Overall risk is determined by the highest risk score for any criterion (modified from Lindley et al. 2007).

Criterion	Risk of extinction		
	High	Moderate	Low
Extinction risk and PVA	> 20% within 20 yrs	> 5% within 100 yrs	< 5% within 100 yrs
	- or any ONE of -	- or any ONE of -	- or ALL of -
Population size[a]	$N_e \leq 50$	$50 < N_e \leq 500$	$N_e > 500$
	- or -	- or -	- or -
	$N \leq 250$	$250 < N \leq 2500$	$N > 2500$
Population decline	Precipitous decline[b]	Chronic decline or depression[c]	No decline apparent or probable
Catastrophe, rate, and effect[d]	Order of magnitude decline within one generation	Smaller but significant decline[e]	Not apparent
Hatchery influence[f]	High	Moderate	Low

a – Census size N can be used if direct estimates of effective size N_e are not available, assuming $N_e/N = 0.2$.
b – Decline within last two generations to annual run size ≤ 500 spawners, or run size > 500 but declining at ≥ 10% per year over the past 10 years. Historically small but stable population not included.
c – Run size has declined to ≤ 500, but now stable.
d – Catastrophes occurring within the last 10 years.
e – Decline < 90% but biologically significant.
f – See Figure 5.3 for assessing hatchery impacts.

Table 5.2. Viability metrics for Sacramento River Winter-run Chinook Salmon ESU populations. Total population size (N) is estimated as the sum of estimated run sizes over the most recent three years. The mean population size ($\hat{S}$) is the average of the estimated run sizes for the most recent three years. Population growth rate (or decline; 10 year trend) is estimated from the slope of log-transformed estimated run sizes. The catastrophic metric (Recent Decline) is the largest year-to-year decline in total population size (N) over the most recent 10 such ratios.

Population	N	$\hat{S}$	10-year trend (95% CI)	Recent Decline (%)
LSNFH winter-run Chinook	645	215.0	0.102 (-0.019, 0.222)	2.7
Sacramento River winter-run Chinook	11125	3708.3	-0.155 (-0.345, 0.034)	67.4

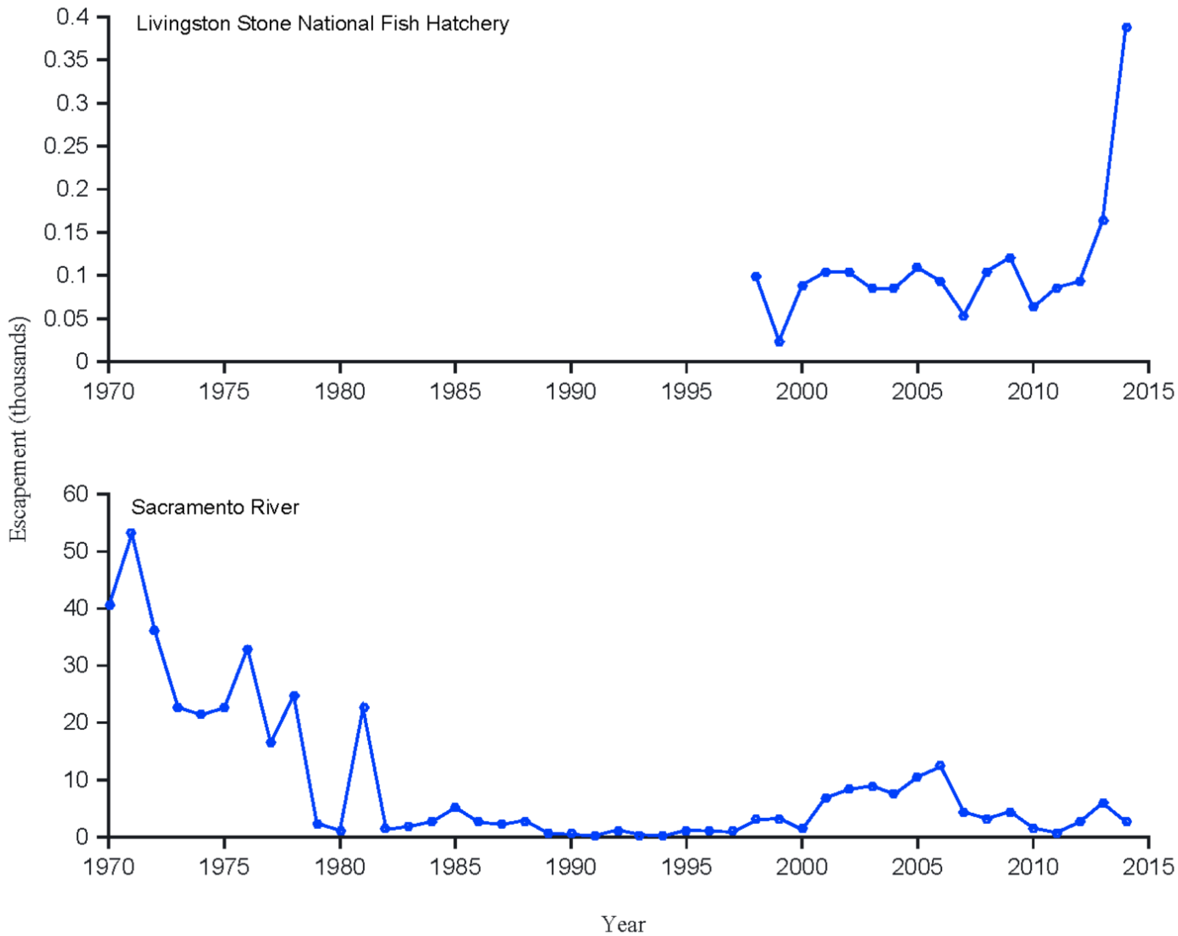

Figure 5.1. Time series of escapement for Sacramento River Winter-run Chinook Salmon used as broodstock at Livingston Stone National Fish Hatchery and Sacramento River mainstem spawners. Estimates for in-river spawners is the average number of adults counted at Red Bluff Diversion Dam and the carcass survey mark-recapture estimates (when available). Note: only mark-recapture estimates are used beginning in 2009; data from Azat (2014).

(approximately four generations) is 13% (SD= ±8%) with the most recent generation at 20% hatchery influence, placing the population at a moderate risk of extinction (Table 5.3; Figure 5.3).

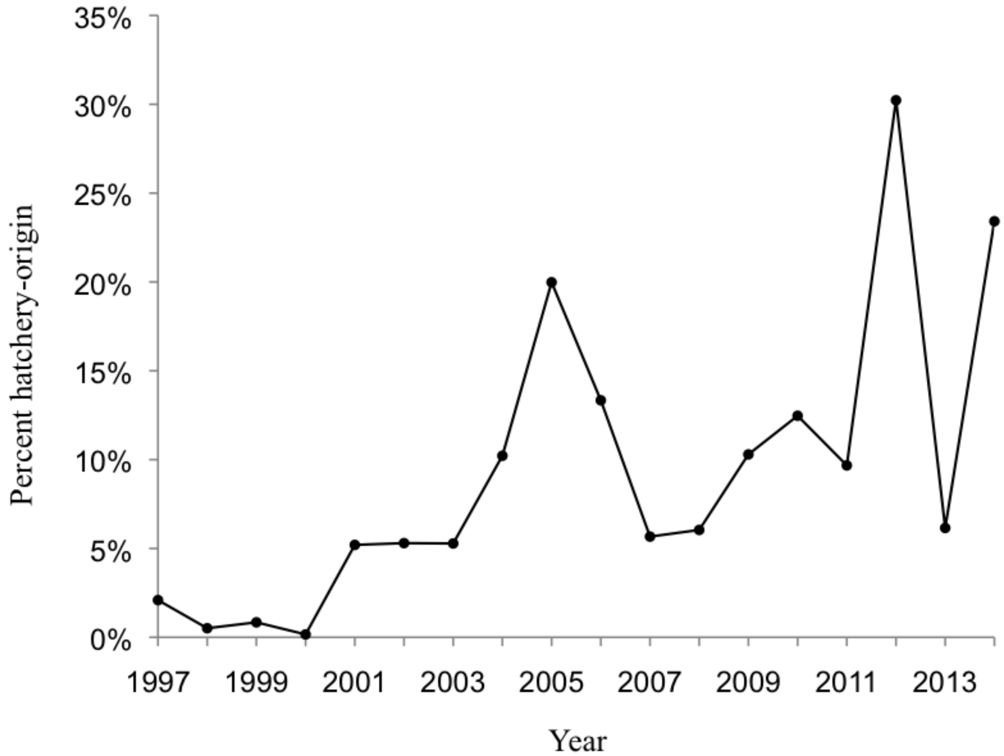

Figure 5.2. Percentage of in-river spawning Sacramento River Winter-run Chinook Salmon that are of hatchery-origin; Data source: Killam 2014.

Table 5.3. Average percentage of hatchery-origin Sacramento River Winter-run Chinook salmon spawners over a varying (cumulative) number of years. One generation (g1) consists of the most recent three years; two generations (g2) the most recent six years; three generations (g3) the most recent nine years; four generations (g4) the most recent 12 years. Data source: Killam 2014).

	g1	g2	g3	g4
Average hatchery influence	20%	15%	13%	13%

Harvest Impacts[12]

Sacramento River Winter-run Chinook Salmon (SRWRC) have a more southerly ocean distribution relative to other California Chinook salmon stocks, and are primarily impacted by fisheries south of Point Arena, California. Sacramento River Winter-run Chinook Salmon age-3 ocean fishery impact rate estimates for the region south of Point Arena (an approximation of the exploitation rate) are currently available for 2000–2013, and have remained relatively stable over this time period, averaging 16% (Figure 5.4). Fisheries in 2008 and 2009 were closed south of Point Arena owing to the collapse of the Sacramento River Fall-run Chinook salmon stock and insufficient data exist for estimating a SRWRC impact rate in 2010. If years 2008–2010 are omitted, the average age-3 impact rate is 19% (PFMC 2015b).There have been several layers of ocean salmon fishery regulations implemented for SRWRC beginning in the early 1990s. For example, a substantial portion of the SRWRC ocean harvest impacts used to occur in February and March recreational fisheries south of Point Arena, but fisheries at that time of the year have been closed since the early 2000s. O'Farrell and Satterthwaite (2015) hindcasted SRWRC age-3 ocean impact rates back to 1978, extending the impact rate time series beyond the range of years where direct estimation is possible (2000–2013). Their results suggest that there were substantial reductions in ocean impact rates prior to 2000 and that the highest impact rates occurred in a period between the mid-1980s and late-1990s.

One component of the Reasonable and Prudent Alternative (RPA) from the 2010 Biological Opinion (NMFS 2010) specified that new fishery management objectives must be established. The implementation of the RPA resulted in the development of a harvest control rule which was first used for ocean fishery management in 2012. That harvest control rule specifies reductions in the age-3 ocean impact rate when the geometric mean number of spawners from the previous three years is reduced (Figure 5.5). The limits to the impact rate imposed by the harvest control rule is an additional control on ocean fisheries which still includes previously existing constraints on fishery opening and closing dates and minimum size limits south of Point Arena. Between 2012 and 2015, the SRWRC harvest control rule has specified maximum allowable forecast impact rates ranging from 12.9% to 19.0%.

What little SRWRC freshwater harvest that existed was essentially eliminated beginning in 2002 when Sacramento River Chinook salmon fishery season openings were adjusted to reduce the temporal overlap with the SRWRC spawning migration and spawning period.

In summary, the available information indicates that the level of SRWRC fishery impacts has not changed appreciably since the 2010 salmon and steelhead viability assessment (Williams et al. 2011), yet there have been additional ocean fishery regulations implemented with the purpose of reducing exploitation of SRWRC when average population size is reduced.

---

[12] Harvest impacts section prepared by Michael O'Farrell

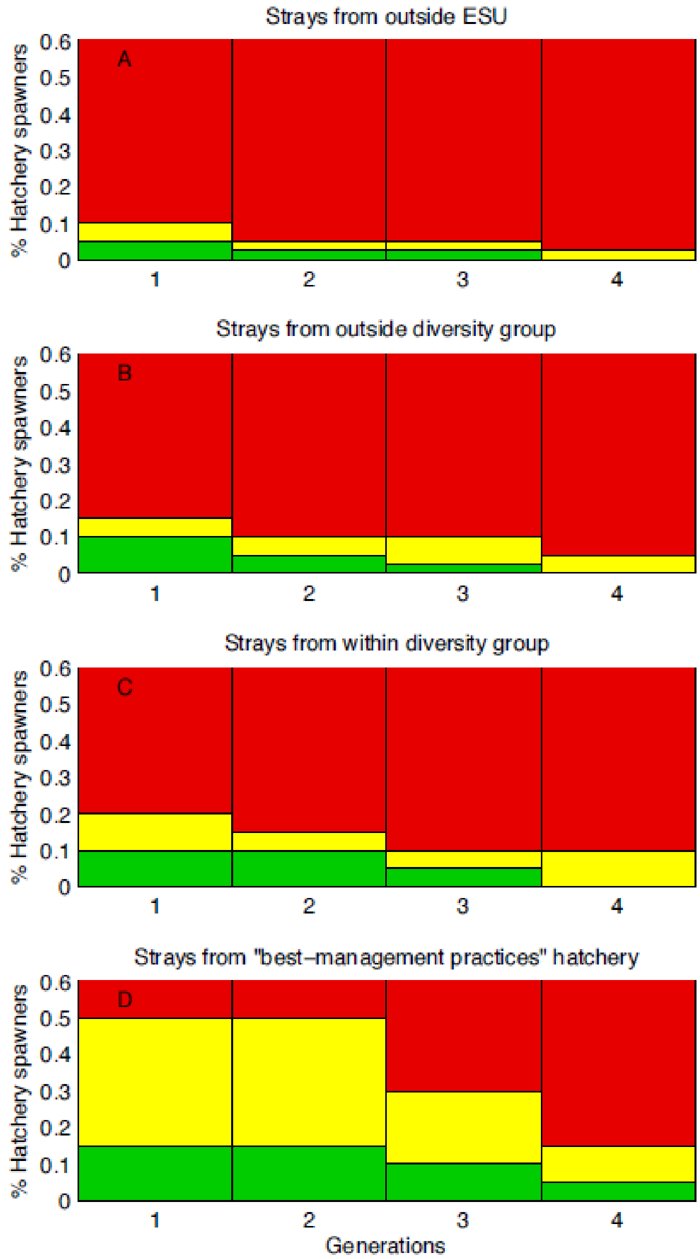

Figure 5.3. Percentage of hatchery-origin spawners and the resulting risk of extinction due to hatchery introgression from different sources of strays over multiple generations for Sacramento River Winter-run Chinook salmon. Low (green), moderate (yellow), and high (red). Model using “best-management practices” was used in the winter-run assessment based on the breeding protocols at the Livingston Stone National Fish Hatchery for Sacramento River Winter-run Chinook Salmon. The group/parameter “strays from outside of ESUs” was used to assess impacts of introgression between Central Valley Spring- and Fall-run Chinook Salmon ESUs at the Feather River Hatchery. Figure reproduced from Lindley et al. (2007).

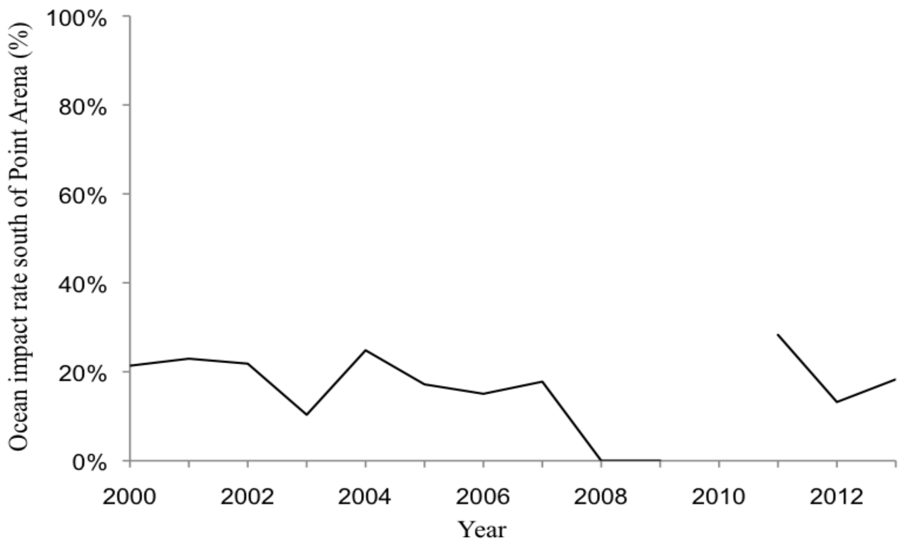

Figure 5.4. Sacramento River Winter-run Chinook Salmon age-3 ocean impact rate (percent) south of Point Arena, California for years 2000–2013. Estimates are sourced from PFMC (2015b). The impact rate could not be estimated in 2010 due to insufficient coded-wire tag recoveries.

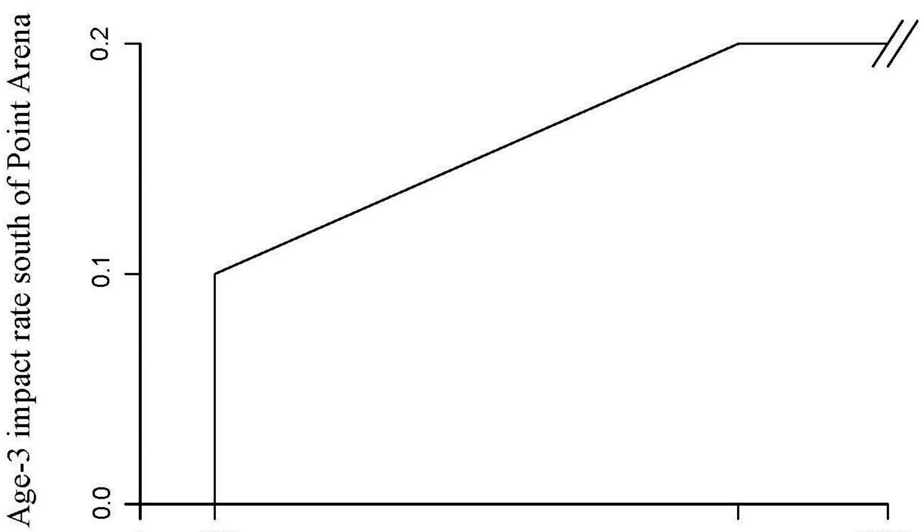

Figure 5.5. Current Sacramento River Winter-run Chinook Salmon harvest control rule. There is no explicit cap on the age-3 impact rate if the three-year geometric mean number of spawners exceeds 5000.

Summary and Conclusions

The overall viability of Sacramento River Winter-run Chinook Salmon has declined since the 2010 viability assessment, with the single spawning population on the mainstem Sacramento River. New information available since Williams et al. (2011) indicates an increased extinction risk to this ESU. The larger influence of the hatchery broodstock in addition to the rate of decline in abundance over the past decade has placed the population at an increased risk of extinction (Table 5.4).

The SRWRC population has declined during recent periods of unfavorable ocean conditions (2005–2006), and droughts (2007–2009) and are expected to continue to be low due to drought conditions in 2012–2015. The low adult returns in 2011 created a potential increase in vulnerability to a year class, yet the progeny from this cohort had relatively high survival resulting in a positive cohort replacement rate (3.5) from this numerically weak cohort (Azat 2014).

Poor early life stage survival during the most recent consecutive drought years of 2012–2015, coupled with poor ocean conditions and hatchery production practices (see Chapter 2) may further impact SRWRC survival-to-adulthood and risk of extinction. Temperature conditions during egg development and fry emergence were suboptimal over the duration of SRWRC rearing in 2014 and 2015 due to reduced cold water storage and subsequent release in/from Shasta Reservoir for this life stage. The egg-to-fry survival estimate for brood year 2014 is 5.9%, which is a significant departure from the average of 24.8% for brood years 1996–2014 measured at RBDD (Poytress et al. 2016). Potential impacts to these cohorts would be observed in viability criteria once adults return in 2015 and beyond.

Water operations can influence the routing of upper Sacramento River-origin water through agricultural fields and can create false attraction cues that cause SRWRC to deviate from the mainstem Sacramento River migration corridor and become stranded in agricultural fields behind flood bypass weirs. SRWRC have been observed to navigate up the Colusa Basin Drain for 40–70 miles before being blocked at weirs delaying and/or preventing successful migration (CALFED 2000, USFWS 2001, USBR and DWR 2012). In 2013, 600+ stranded adult SRWRC and CVSRC were observed, with a total of 312 adults relocated to the mainstem Sacramento River or the Livingston Stone National Fish Hatchery for use as broodstock (Killam et al. 2014). It is likely that survival for rescued adults that were stranded in the Colusa Basin Drain was low and that not all stranded adults were rescued. Thus, the loss of adults due to stranding prior to spawning can be demographically costly to the population.

The SRWRC ESU is likely at a lower extinction risk with a sustainable LSNFH population and naturally spawning population than it would be with just a single naturally spawning population, at least in the near-term. Yet, reliance on production from LSNFH and potential introgression between natural-origin SRWRC is increasing (Figure 5.2). In an attempt to prevent the loss of SRWRC cohorts during the 2013–2015 prolonged drought, a greater number of spawners were brought into the LSNFH as broodstock (Figure 5.1). The hatchery also produced and released three times as many juveniles. Thus, in years where mortality of natural-origin fish may be particularly high and LSNFH production is significantly increased, the contribution of LSNFH-origin fish to the

Table 5.4. Summary of Sacramento River Winter-run Chinook salmon extinction risk by population criteria described in Lindley et al. (2007) for the 2010 and 2015 review periods. Overall risk is determined by the highest risk score for any criterion.

	2010 Status Review	2015 Status Review
Population size	Low risk	Low risk
Population decline	Low risk	Moderate risk
Catastrophe, rate, and effect	Low risk	Low risk
Hatchery influence	Low risk	Moderate risk

returning adult spawners may elevate the overall risk of extinction of SRWRC due to genetic impacts from the hatchery. Potential impacts would manifest in viability criteria evaluations in escapement from the year 2016 and beyond, unless hatchery introgression is minimized through active adult management on the spawning grounds. The use of adult segregation weirs to manage gene flow between natural- and hatchery-origin fish in rivers is commonly conducted in Oregon and Washington to minimize impacts of hatchery fish on the genetic integrity of the overall population (HSRG 2014).

The viability of the SRWRC ESU will be improved by re-establishing winter-run Chinook salmon in their historical spawning and rearing habitat. Projects to reintroduce SRWRC into Battle Creek and upstream from Shasta Reservoir are in the planning phases, and if successful, would significantly benefit SRWRC. Genetic management plans will be critical for conserving the long-term genetic integrity of SRWRC, the success of the reintroduction efforts, and achieving a low-extinction risk to the portion of the population downstream Shasta Dam.

Lastly, the development and implementation of quantitative modeling tools that link water project operations, temperature management, and habitat restoration actions to SRWRC population dynamics will greatly improve our ability to make science-informed management decisions (Hendrix et al. 2014; Caldwell et al. 2015).

5.2 Central Valley Spring-run Chinook Salmon ESU

ESU Boundary Delineation

The Central Valley Spring-run Chinook Salmon (CVSRC) ESU includes spring-run Chinook salmon populations spawning in the Sacramento River and its tributaries, and spring-run Chinook salmon in the Feather River Hatchery (FRH).

The San Joaquin Delta and entire watershed is excluded as Critical Habitat and its populations considered extirpated (64 FR 50394; 70 FR 52488). Information on the presence of fish exhibiting spring-run behavior in San Joaquin River tributaries is provided and may represent passive recolonization of CVSRC into the San Joaquin River Basin. Thus, there is value in continuing to monitor these populations to evaluate the extent to which populations in the San Joaquin River tributaries warrant inclusion in the ESU boundary. No new information suggests that the boundary of this ESU should change or that its status as an ESU should change.

Summary of Previous Assessments

Williams et al. (2011) found that the viability of the ESU had probably deteriorated since the 2005 review (Good et al. 2005). Williams et al. (2011) reported improvements, evident in the viability of two populations, although these population level improvements were not enough to warrant a downgrading of the ESU extinction risk and there had been an overall increase in extinction risk to the ESU since the review by Good et al. (2005).

Brief Review of TRT Documents and Previous Findings

The TRT delineated 18 or 19 independent populations of CVSRC, along with a number of smaller dependent populations, and four diversity groups (Lindley et al. 2004). Of these 18 populations, only three are extant (Mill, Deer, and Butte creeks on the upper Sacramento River) and they represent only the Northern Sierra Nevada diversity group. All populations in the Basalt and Porous Lava group and the Southern Sierra Nevada group were extirpated, and only a few dependent populations persist in the Coast Range group. Using data through 2005 and the criteria in Table 5.1, Lindley et al. (2007) found that the Mill Creek, Deer Creek, and Butte Creek populations were at or near low risk of extinction. However in 2010, declines in abundance placed Mill and Deer Creek populations at a high risk of extinction due to their rates of decline, and in the case of Deer Creek, also the level of escapement. The ESU as a whole was not considered viable because there were no extant populations in the three other diversity groups. In addition, Mill, Deer and Butte creeks are close together geographically, decreasing the independence of their extinction risks due to catastrophic disturbance (Lindley et al. 2007).

New Data and Updated Analyses

Figure 5.6 shows the escapement of CVSRC to various Central Valley streams, and Table 5.5 shows abundance and trend statistics related to viability criteria. All independent populations (Battle, Deer, Mill, and Butte creek populations) show larger total population sizes (*N*) and mean escapement ($\hat{S}$) than the previous assessment in 2010. New data for the Yuba River suggests a low extinction risk based on population size. The Butte Creek population remains at a low risk, while Deer Creek and Mill Creek populations have shown improvements from a high risk to a moderate risk with population sizes approaching low risk abundance thresholds. In particular, Butte Creek spring-run Chinook salmon appear to be trending in a positive direction with improvements to all viability criteria since the previous 2010 assessment. Butte Creek's total population size is 20,169 which is twice the 2010 estimate and remains the largest CVSRC population (Table 5.5).

The majority of CVSRC populations are still exhibiting declines in run sizes over time, with the exception of Clear Creek, Battle Creek, and Butte Creek populations which have positive point estimates of population growth (Table 5.5). In particular, CVSRC appear to be repopulating Battle Creek, home to a historical independent population in the Basalt and Porous Lava diversity group that was extirpated for many decades. Abundance has increased 18% over the last decade (N= 1836) that qualify it for a moderate extinction risk score and trending towards a low-risk threshold of 2500 fish (Lindley et al. 2007). Similarly, the CVSRC population in Clear Creek has been increasing, although Lindley et al. (2004) classified this population as a dependent population, and thus it is not expected to exceed the low-risk population threshold of 2500 fish. While the viability of independent populations has generally improved, the majority of dependent populations have declined in abundance over the last decade (Table 5.5). Recent declines have been significant and almost qualify as catastrophes under the criteria (>90% decline) of Lindley et al. (2007) with the dependent Antelope Creek and Cottonwood Creek populations, and the independent Deer Creek population experiencing recent declines of >80% in one generation (Table 5.5).

Hatchery introgression between Feather River spring- and fall-run Chinook salmon ESUs in the breeding program at the FRH compromises the long-term genetic integrity of the spring-run Chinook salmon population on the Feather River and poses a high extinction risk. Coded-wire tag returns confirm that fish identified as FRH spring-run Chinook salmon are intermixed at the hatchery with those identified as fall-run Chinook salmon (Hedgecock et al. 2001; California HSRG 2012). In 2011, 40% of the FRH spring-run Chinook salmon broodstock was comprised of fall-run individuals (Palmer-Zwahlen and Kormos 2013). Based on the moderate extinction risk threshold for gene-flow for one generation between ESUs (< 10%), it places the FRH and naturally spawned CVSRC in the Feather River at a high risk of extinction (Kormos et al. 2012; Palmer-Zwahlen and Kormos 2013; Figure 5.3).

The majority of the FRH spring-run Chinook salmon broodstock and in-river spawning population on the Feather River are first generation hatchery-produced fish (Kormos et al. 2012; Palmer-Zwahlen and Kormos 2013). The proportion of natural-origin fish in the broodstock is estimated to be 18% (2010) and 6% (2011) (Kormos et al. 2012; Palmer-Zwahlen and Kormos 2013). Thus, the minimum criteria of >10% of natural-origin fish

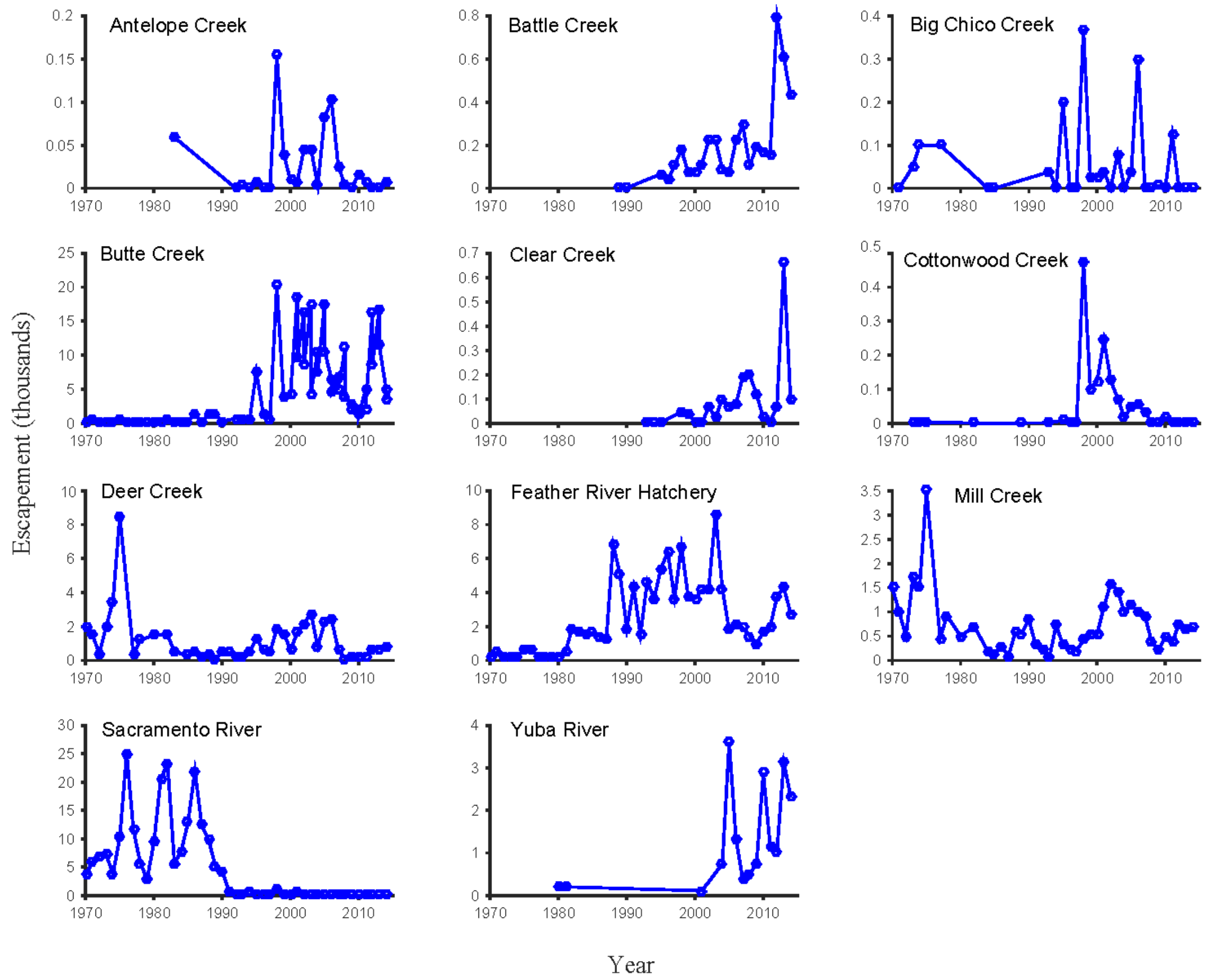

Figure 5.6. Escapement for Central Valley Spring-run Chinook Salmon populations over time in thousands of fish. Note: Beginning in 2009, Red Bluff Diversion Dam estimates of spring-run Chinook salmon in the Upper Sacramento River were no longer available.

in the broodstock is not being met annually (California HSRG 2012). The proportion of hatchery-origin spring- or fall-run contributing to the natural spawning spring-run Chinook salmon population on the Feather River remains unknown due to overlap in the spawn timing (and thus coded-wire tag recoveries in carcass surveys) of spring- and fall-run Chinook salmon. However, the hatchery component is likely to be high. For example, 78% and 90% of spawners in the 2010–2011 spring-run/fall run carcass survey were estimated to be from the FRH respectively (Kormos et al. 2012; Palmer-Zwahlen and Kormos 2013).

Table 5.5. Viability metrics for Central Valley Spring-run Chinook Salmon ESU populations. Total population size (*N*) is estimated as the sum of estimated run sizes over the most recent three years for independent populations (**bold**) and dependent populations. The mean population size ($\hat{S}$) is the average of the estimated run sizes for the most recent three years. Population growth rate (or decline; 10-year trend) is estimated from the slope of log-transformed estimated run sizes. The catastrophic metric (Recent Decline) is the largest year-to-year decline in total population size (*N*) over the most recent 10 such ratios.

Population	*N*	$\hat{S}$	10-yr trend (95% CI)	Recent decline (%)
Antelope Creek	8	2.7	-0.375 (-0.706, -0.045)	87.8
Battle Creek	1836	612.0	0.176 (0.033, 0.319)	9.0
Big Chico Creek	0	0.0	-0.358 (-0.880, 0.165)	60.7
Butte Creek	20169	6723.0	0.353 (-0.061, 0.768)	15.7
Clear Creek	822	274.0	0.010 (-0.311, 0.330)	63.3
Cottonwood Creek	4.0	1.3	-0.343 (-0.672, -0.013)	87.5
Deer Creek	2272	757.3	-0.089 (-0.337, 0.159)	83.8
Feather River Hatchery	10808	3602.7	0.082 (-0.015, 0.179)	17.1
Mill Creek	2091	697.0	-0.049 (-0.183, 0.086)	58.0
Sacramento River[a]	-	-	-	-
Yuba River	6515	2170.7	0.67 (-0.138, 0.272)	9.0

a – Beginning in 2009, estimates of spawning escapement of Upper Sacramento River spring-run Chinook were no longer monitored. Historically, this estimate was derived by the total Red Bluff Diversion Dam (RBDD) counts minus the spring-run Chinook salmon adult counts in the upper Sacramento River tributaries. Beginning in 2009, RBDD gates were partially operated in the up position and in 2012 they were entirely removed, and thus spring-run estimates are no longer available.

The spring-run Chinook salmon population in the Yuba River is at a low risk of extinction based on total population size (N=6,512) yet a high risk due to a conservative estimate of the percentage of hatchery spawners (10 year average = 19%) and likely introgression between fall- and spring-run individuals (RMT 2015). The abundance of spring-run Chinook salmon passing Daguerre Point Dam on the Yuba River was not evaluated in the previous 2010 viability report due to difficulties in differentiating spring-run from fall-run individuals in the existing monitoring. Currently, a video camera is used to count fish moving upstream in the ladders at Daguerre Point Dam. A method for delineating a temporal window for passage of spring- and fall-run passage has been developed and thus counts of spring-run Chinook salmon and fall-run Chinook salmon are reported for 2004–2011 (RMT 2015). Additionally, data on the percentage of adults with and without adipose fins based on silhouettes in the video are reported for 2004–2011(RMT 2015).

Genetic studies suggest that hybridization between FRH spring-run Chinook salmon and other Central Valley Chinook salmon runs has not occurred, where evaluated. For example, where FRH CVSRC strayed extensively, the effect is not apparent in the genetic structure described by microsatellite markers for Central Valley Spring-run Chinook salmon runs in Mill, Deer and Butte creeks, or on winter-run and late-fall-run Chinook salmon that spawn in the mainstem Sacramento River (Banks et al. 2000). These findings are consistent with the generally low straying rates estimated by recovery of coded-wire tags (Kormos et al. 2012; Palmer-Zwahlen and Kormos 2013). However, FRH CVSRC adults have been recovered in other Central Valley spring- and fall-run Chinook salmon populations outside of the Feather River. Feather River Hatchery spring-run Chinook salmon smolts released into the San Francisco Bay pose greater genetic risk to other Central Valley Chinook salmon populations than those released in-river at the hatchery based on their greater stray-rates (Kormos et al. 2012; Palmer-Zwahlen and Kormos 2013). On Clear Creek, 0%-5% of spring-run Chinook salmon carcasses above the spring-run segregation weir in 2010–2013 were from the FRH (USFWS 2014). In 2010 as many as 29% of the CVSRC were estimated to have originated from FRH on Battle Creek (USFWS 2014). A significant number of FRH spring-run Chinook salmon strays have been observed in 2015 in the Upper Sacramento River at the Keswick Dam trap (N=114) and could be interbreeding with natural-origin spring- or fall-run Chinook salmon(J. Rueth, USFWS, personal communication). Prolonged influx of FRH spring-run Chinook salmon strays to other spring-run Chinook salmon populations even at levels <1% is undesirable and can cause the receiving population to shift to a moderate risk after four generations of such impact (Lindley et al. 2007; Figure 5.3). Additional information on the incidence of FRH spring-run Chinook salmon straying is desirable to more accurately estimate the extent to which spawning and introgression is occurring between fall- and spring-run Chinook salmon populations outside of the Feather River.

For many decades, CVSRC were considered extirpated from the Southern Sierra Nevada diversity group in the San Joaquin River Basin, despite their historical numerical dominance in the Basin (Fry 1961; Fisher 1994). More recently, there have been reports of adult Chinook salmon returning in February through June to San Joaquin River tributaries, including the Mokelumne, Stanislaus, and Tuolumne rivers (Workman 2003; Franks 2012; Guignard 2015). These spring-running adults have been observed in several years and exhibit typical spring-run life-history characteristics, such as returning to tributaries during the springtime, over-summering in deep pools, and spawning in early fall (Workman 2003; Franks 2011; Guignard 2015). For example, 114 adults were counted using a video weir on the Stanislaus River between February and June in 2013 with only 7 individuals observed without adipose fins (Guignard 2015). Since all hatchery-origin CVSRC have their adipose fins removed, these data suggest the vast majority of the adult spring-run Chinook salmon were not strays from the FRH. It is possible that they are unmarked fall-run Chinook salmon hatchery adults that strayed to the Stanislaus River from the Feather River Hatchery, Coleman National Fish Hatchery, Nimbus River Fish Hatchery, Mokelumne River Hatchery, or Merced River Hatchery. The extent to which these phenotypic spring-run Chinook salmon have a similar genetic lineage as other extant spring-run Chinook salmon populations and that they stray each generation from the Sacramento River Basin remains unknown and is the source of on-going research. It is conceivable that progeny from adult spring-run Chinook salmon return to their natal tributaries on the San Joaquin River and thus represent early stages of a recolonization process trending towards a self-sustaining population. Juveniles expressing atypical fall-run outmigration behavior, more characteristic of spring-run (e.g., yearlings) have also been observed on the Mokelumne, Tuolomne, and Stanislaus rivers (Fuller 2008; Watry et al. 2012; Bilski et al. 2013). In addition, in 2014, a reintroduction program was initiated as part of the San Joaquin River Restoration Program, and 54,000 juvenile spring-run Chinook salmon were released into the river. Successful reestablishment of CVSRC into multiple populations in the Southern Sierra Nevada Group would significantly increase their spatial diversity and decrease their risk of extinction.

Harvest Impacts[13]

Attempts have been made (Grover et al. 2004) to estimate Central Valley Spring-run Chinook (CVSRC) Salmon ESU ocean fishery exploitation rates using coded-wire tag recoveries from natural origin Butte Creek fish, but due to the low number of recoveries the uncertainty of these estimates is too high for them to be of value. However, because CVSRC have a relatively broad ocean distribution from central California to Cape Falcon, Oregon, that is similar to that of Central Valley Fall-run Chinook (CVFRC) salmon, trends in the CVFRC ocean harvest rate may provide a reasonable proxy for trends in the CVSRC ocean harvest rate. While the CVFRC ocean harvest rate can provide information on trends in CVSRC fishing mortality, it is possible that CVSRC experiences lower overall fishing mortality. If maturation rates are similar between CVSRC and CVFRC, the ocean exploitation rate on CVSRC would be lower than CVFRC in the last year of life because spring-run Chinook salmon escape ocean fisheries in the spring, prior to the most extensive ocean salmon fisheries in summer.

The CVFRC ocean harvest rate index peaked in the late 1980s and early 1990s, but then declined (Figure 5.7). With the closure of nearly all Chinook salmon ocean fisheries south of Cape Falcon in 2008 and 2009, the index dropped to 6% and 1%, respectively. While ocean fisheries resumed in 2010, commercial fishing opportunity was severely constrained, particularly off California, resulting in a harvest rate index of 16%. Since 2011, ocean salmon fisheries in California and Oregon have had more typical levels of fishing opportunity. The average CVFRC ocean harvest rate between 2011 and 2014 is 45% which is generally similar to levels observed between the late 1990s and 2007.

The CVSRC spawning migration largely concludes before the mid- to late-summer opening of freshwater salmon fisheries in the Sacramento Basin, and salmon fishing is prohibited altogether on Butte, Deer, and Mill creeks, indicating that CVSRC river fishery impacts are relatively minor.

[13] Harvest impacts section prepared by Michael O'Farrell

Figure 5.7. Central Valley Fall-run Chinook (CVFRC) Salmon ocean harvest index for years 1983 – 2014. The harvest rate index is computed from estimates presented in Table II-1 from PFMC (2015b).

In summary, the available information indicates that the level of CVSRC fishery impacts has not changed appreciably since the 2010 salmon and steelhead assessments (Williams et al. 2011).

Summary and Conclusions

Central Valley-wide, the viability of CVSRC has probably improved on balance since the 2010 viability assessment with improvements to Mill Creek and Deer Creek populations changing from high-risk to moderate-risk of extinction. In fact, total abundance of CVSRC for the Sacramento River watershed in 2014 (not including the FRH or Feather River but with the addition of Yuba River spring-run Chinook salmon) is 45,215, close to the decadal high of 55,827 (2004) and a factor of approximately four times higher than the decadal low of 12,207 which occurred as recently as 2012 (Azat 2014; RMT 2015). The Central Valley-wide abundance is driven largely by the annual variation in Butte Creek returns. Butte Creek remains at low risk, and all viability metrics are trending in a positive direction. The Butte Creek spring-run Chinook salmon population has increased in part due to extensive habitat restoration and the accessibility of floodplain habitat in the Sutter-Butte Bypass for juvenile rearing in the majority of years. Most dependent spring-run Chinook salmon populations have been experiencing continued and somewhat drastic declines. Counteracting these developments, CVSRC have repopulated Battle and

Clear creeks where they were once extirpated and have increased in abundance over the last decade, reaching levels of abundance that place these populations at moderate extinction risk. In the case of Clear Creek, the majority of fish spawning there are of natural origin (96%) suggesting local production may be promoting a self-sustaining population without significant hatchery supplementation (Kormos et al. 2012; Palmer-Zwahlen and Kormos 2013).

Central Valley Spring-run Chinook salmon populations have experienced a series of droughts over the past decade. From 2007–2009 and 2012–2015, the Central Valley experienced drought conditions and low river and stream discharges, which are generally associated with lower survival of Chinook salmon. The impacts of the recent drought series and warm ocean conditions on the juvenile life stage (see Chapter 2 of this assessment) will not be fully realized by the viability metrics until they manifest in potential low run sizes in 2015–2018.

The recent drought has impacted CVSRC adults on Butte Creek, which have experienced lethal temperatures in traditional and non-traditional holding habitat during the summer. A large number of adults (903 and 232) were estimated to have died prior to spawning in the 2013 and 2014 drought respectively (Garman 2015). Pre-spawn mortality was also observed during the 2007–2009 drought with an estimate of 1,054 adults dying before spawning in 2008 (Garman 2015). In 2015, late-arriving adults in the vicinity of the City of Chico experienced exceptionally warm June air temperatures coupled with the Pacific Gas and Electric flume shutdown resulting in a fish die-off. Thus, while the independent CVSRC populations have generally improved since 2010 and are considered at moderate and low risk of extinction, the viability of CVSRC populations are not likely improvement over the next three years due to likely unfavorable hydroclimatic regimes.

Current introgression between fall- and spring-run Chinook salmon in the FRH breeding program and straying of FRH spring-run Chinook salmon to other non-hatchery spring-run Chinook salmon populations could compromise the genetic integrity of spring-run Chinook salmon populations. Off-site releases of FRH spring-run Chinook salmon has resulted in increased straying of hatchery fish into other spring-run Chinook salmon populations and if continued could result in a moderate risk of extinction to other spring-run Chinook salmon populations. However, beginning in 2014, and expected to continue, the FRH has released spring-run Chinook salmon juveniles into the Feather River rather than releasing them in the San Francisco Bay, which is hypothesized to reduce straying (California HSRG 2012).

At the ESU level, the spatial diversity within the CVSRC ESU is increasing and spring-run are present (albeit at low numbers in some cases) in all diversity groups. The recolonization of CVSRC to Battle Creek and increasing abundance of CVSRC on Clear Creek is benefiting the viability of CVSRC. Similarly, the reappearance of phenotypic spring-run to the San Joaquin River tributaries may be the beginning of natural recolonization processes in rivers where they were once extirpated. Active reintroduction efforts on the Yuba River and below Friant Dam on the mainstem San Joaquin River show promise and will be necessary to make the ESU viable. The CVSRC ESU is trending in a positive direction towards achieving at least two populations in each of the four historical diversity groups necessary for recovery with the Northern Sierra Nevada region necessitating four populations (NMFS 2014b).

The viability of the CVSRC ESU has likely improved since the 2010 viability assessment. Largest improvements are due to the increase in spatial diversity with historically extirpated populations trending in the positive direction. Improvements, evident in the moderate and low risk of extinction of the three independent populations, are certainly not enough to warrant a downgrading of the ESU extinction risk. The recent catastrophic declines of many of the dependent populations, high pre-spawn mortality during the 2012–2015 drought, uncertain juvenile survival due to the drought and variable ocean conditions, as well as the level of straying of FRH spring-run Chinook salmon to other spring-run Chinook salmon populations are all causes for concern for the long-term viability of the CVSRC ESU.

5.3 California Central Valley Steelhead DPS

DPS Boundary Delineation

This Distinct Population Segment (DPS) includes steelhead populations spawning in the Sacramento and San Joaquin rivers and their tributaries. Hatchery stocks within the DPS include Coleman National Fish Hatchery (CNFH) and Feather River Hatchery (FRH); steelhead in the Nimbus Hatchery (NH) and Mokelumne River Hatchery (MRH) are currently excluded from the DPS. New genetic analysis show that the steelhead stock currently propagated in the Mokelumne River Hatchery is genetically similar to the steelhead broodstock in the FRH (Pearse and Garza 2015), consistent with documentation on the recent transfers of eggs from the FRH for broodstock at the MRH. The NH steelhead remain genetically divergent from the Central Valley DPS lineages, consistent with their founding from coastal steelhead stocks, and remain excluded from the DPS (Pearse and Garza 2015). Thus, we recommend a change in boundary delineation, the boundary of the Central Valley DPS should be modified to include steelhead from the Mokelumne River Hatchery.

Summary of Previous Assessments

Good et al. (2005) found that California Central Valley (CCV) Steelhead DPS was in danger of extinction, with a minority of the BRT viewing the DPS was likely to become endangered. The BRT's major concerns were the low abundance of naturally produced anadromous fish at the DPS (considered an ESU at the time of the review) level, the lack of population-level abundance data, and the lack of any information to suggest that the monotonic decline in steelhead abundance evident from 1967–1993 dams counts had stopped. Williams et al. (2011) reported that the viability of this steelhead DPS had worsened since the 2005 review when Good et al. (2005) concluded that the DPS was in danger of extinction.

Brief Review of TRT Documents and Previous Findings

The Central Valley domain Technical Recovery Team delineated more than 80 independent populations of Central Valley steelhead, along with a number of smaller dependent populations. Many of these historical populations are entirely above impassable barriers and may persist as non-anadromous or adfluvial rainbow trout, although they are presently not considered part of the DPS. Impassable dams also block many populations from reaching significant portions of their historical spawning and rearing habitat.

Lindley et al. (2007) developed viability criteria for steelhead, summarized in Table 5.1. Using data through 2005, Lindley et al. (2007) found that data were insufficient to determine the viability of any of the naturally spawning populations of Central Valley steelhead, except for those spawning in rivers adjacent to hatcheries, which were likely to be at high risk of extinction due to extensive spawning of hatchery-origin fish in natural areas. However from 2000–2010, run size data from Battle Creek, which is the best population-level data available for steelhead, suggested a 17% decline per year, placing the population in a high extinction risk category. The proportion of hatchery-origin fish in the Battle Creek returns averaged 29% over the 2002–2010 period, elevating the level of hatchery influence to a moderate risk of extinction. Lastly, the Chipps Island midwater trawl dataset of USFWS indicated that the decline in natural production of steelhead had continued unabated through 2010, with the proportion of adipose fin-clipped steelhead reaching 95%.

New Data and Updated Analyses

Population trend data remain extremely limited for the CCV-Steelhead ESU. The total populations on Battle Creek, CNFH, and FRH have significantly increased since the 2010 assessment with all three populations showing positive population growth estimates over the last decade (Figure 5.8; Table 5.6). Additional data are now available for the American River and Clear Creek steelhead populations and are based on redd counts. Thus, steelhead populations on the American River and Clear Creek are evaluated for the first time using the viability criteria recognizing that some redds in Clear Creek may be from non-anadromous *O. mykiss* (Figure 5.8; Table 5.6).

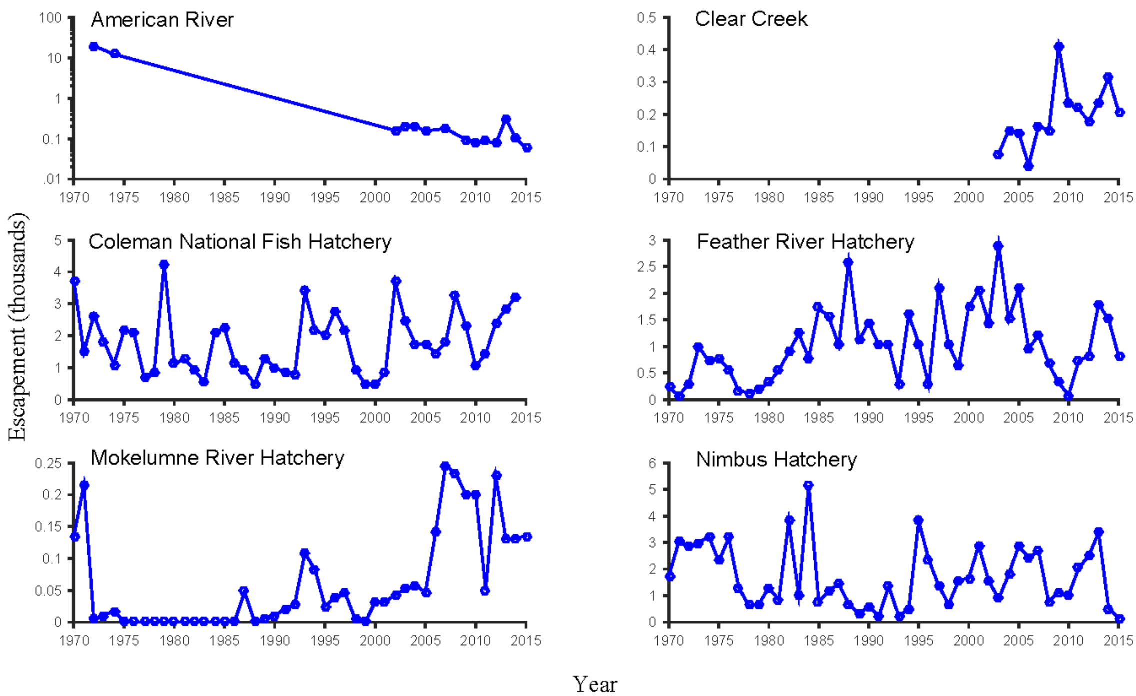

Figure 5.8. Time series of escapement for California Central Valley Steelhead populations in thousands of fish. Note that the y-axis of plot for American River steelhead is on a logarithmic scale.

The best population-level data come from Battle Creek, where CNFH operates a weir. California Central Valley steelhead have been identified as a priority species for restoration in Battle Creek above the weir as part of the Battle Creek Salmon and Steelhead Restoration Project (BCSSRP) and also are produced at CNFH. The Battle Creek watershed is thought to have high potential to support a viable independent population of CCV steelhead within the Basalt and Porous Lava diversity group (NMFS 2009a). In 2002, 2000 steelhead were passed above the weir into the BCSSRP area to spawn naturally in-river. However, prior to 2003, it was not possible to differentiate all hatchery- and natural-origin steelhead, since not all juvenile hatchery fish were adipose fin-clipped and thus a large fraction of these individuals were likely from CNFH (California HSRG 2014). In recent years, so few natural origin steelhead returned to Battle Creek, that beginning in 2009 CNFH was operated as a segregation hatchery with only hatchery steelhead used in the breeding protocols, and only natural origin steelhead passed upstream of the weir into the BCSSRP area (California HSRG 2010). Between 2012 and 2014, the total population of natural-origin adults > 17 inches (size threshold identified for anadromous *O. mykiss* at CNFH; Donohoe and Null 2013) passing the weir was 510 with an average run size of 170 adults (USFWS 2015). The low abundance of natural-

Table 5.6. Viability metrics for California Central Valley steelhead populations. Total population size (N) is estimated as the sum of estimated run sizes over the most recent three. The mean population size ($\hat{S}$) is the average of the estimated run sizes for the most recent three years. Population growth rate (or decline; 10-year trend) is estimated from the slope of log-transformed estimated run sizes. The catastrophic metric (Recent Decline) is the largest year-to-year decline in total population size (N) over the most recent 10 such ratios.

Population	N	$\hat{S}$	10-yr trend (95% CI)	Recent decline (%)
American River[a]	472	157.3	-0.062 (-0.164, 0.039)	45.8
Clear Creek[a]	761	253.7	0.111 (-0.021, 0.244)	9.5
Coleman National Fish Hatchery	8461	2820.3	0.051 (-0.043, 0.146)	18.4
Feather River Hatchery[b]	4119	1373.0	0.061 (-0.171, 0.292)	38.3
Mokelumne River Hatchery	398	132.7	-0.051 (-0.169, 0.067)	30.5
Nimbus Hatchery	4052	1350.7	-0.155 (-0.378, 0.067)	4.5

a – American River and Clear Creek steelhead data are derived from redd counts. Some redds may be from non-anadromous *O.mykiss*.
b – Feather River Hatchery numbers include repeat spawners (fish returning to the hatchery multiple times in a single year). These findings based on recent tagging studies suggest hatchery return numbers are likely slightly inflated.

origin steelhead places it in the moderate extinction risk category, albeit with lower hatchery influence than the previous 2010 assessment. Various management options and potential consequences are currently being evaluated to ensure that natural-origin steelhead that could spawn upstream in the BCSSRP area have an opportunity to reproduce in the wild, while also considering the value of integrating "wild" genes back into CNFH hatchery production to minimize impacts of domestication on both the hatchery- and natural-origin steelhead populations. It is difficult to assess the impact of the CNFH segregation on hatchery steelhead, as little is known about the extent to which CNFH steelhead ascend the segregation weir during high flows and spawn with natural-origin steelhead in the BCSSRP area upstream or with natural-origin steelhead downstream of the weir. In general it requires less influx of hatchery-origin fish from segregated hatcheries than from integrated hatcheries into naturally spawning populations to have significant genetic impacts (California HSRG 2014).

The total population on Clear Creek has increased since it was first estimated in 2003, reaching a total population size of 761, estimated by the number of redds counted and increasing 11% per year over the past decade (Figure 5.8; Table 5.6). American River steelhead had a precipitous decline since 2003, resulting in a moderate risk of extinction based on current total population size estimated by redd surveys. It should be noted that a significant proportion of steelhead redds on the American River are made by NH steelhead, which are not part of the DPS, but are also showing a 15% decline over the last decade.

The NH broodstock remains a high threat to the viability of steelhead populations in the Central Valley. The NH broodstock is not included in the DPS because they are genetically divergent from the Central Valley DPS lineages, having been founded from Eel and Mad River stocks (Pearse and Garza 2015). Thus, potential straying of NH broodstock and continued introgression with natural-origin American River steelhead poses a risk to the overall DPS (California HSRG 2012).

Zimmerman et al. (2009) found that the progeny of anadromous females were present at all eight Central Valley populations evaluated using otolith reconstructions, but the proportion varied among sites (0.04–0.74) and was particularly low for San Joaquin River populations. Data on the presence and numbers of adult steelhead in San Joaquin River tributaries are increasing with the installation of video weirs on the Mokelumne, Stanislaus, and Tuolumne rivers during adult steelhead migration. The numbers of natural-origin adult steelhead remains low, with a high hatchery influence, placing the populations in the San Joaquin River tributaries (Southern Sierra Nevada diversity group) at a high risk of extinction. The annual number of adult steelhead counted moving upstream through the Stanislaus River weir ranged from 1–17 during 2005 to 2008 and 8–32 during 2011 to 2014 (Ford and Kirihara 2010; Fuller 2015.). Thirteen to fifty percent of those fish were identified as hatchery fish having clipped adipose fins, placing the Stanislaus River population at a high risk of extinction based on low numbers and high hatchery influence (Ford and Kirihara 2010; Fuller 2015). The Mokelumne River is also at a high risk of extinction with 92%–96% of adult steelhead at the Woodbridge Dam video identified as hatchery steelhead and with only 3–10 natural-origin steelhead returning to the Mokelumne River each year from 2010–2013 (EBMUD 2011, 2012, 2013).

Steelhead survival from Mossdale on the San Joaquin River to Chipps Island (Delta exit) ranged from 25%–75% in 2010 and 2011 based on acoustic telemetry studies (Buchanan 2013). These survival estimates are significantly higher that what has been observed for fall-run Chinook salmon released in the same location and under similar conditions (SJRGA 2011). The relatively high survival of steelhead is thought to be due to the larger size between the species at release. It is unclear the extent to which naturally produced steelhead experience similar survival rates as the hatchery experimental release groups. In fact, evidence from Chipps Island midwater trawl sampling by USFWS suggests either natural steelhead production and/or survival to Delta exit is very low. The Chipps Island midwater trawl data provide information on the trend in abundance for the CCV Steelhead DPS as a whole. Updated through 2013, the trawl data indicate that the production of natural-origin steelhead remains very low relative to hatchery production (Figure 5.9). Catch-per-unit-effort has fluctuated but remained level over the past decade, but the proportion of the catch that is adipose fin-clipped (100% of hatchery steelhead production have been adipose fin-clipped starting in 1998) has risen steadily, exceeding 90% in recent years, reaching 95% in 2010, and remaining very high through 2013. Because hatchery releases have been fairly constant, this implies that natural production of juvenile steelhead has been falling.

Harvest Impacts[14]

Ocean harvest of steelhead is extremely rare, and is in particular an insignificant source of mortality for the CCV-Steelhead DPS. Insufficient data are available to estimate CCV-Steelhead freshwater exploitation rates directly, though exploitation rates are likely relatively low given that retention of natural-origin steelhead is prohibited. Fishing effort estimates based on angler self-report cards, available from 2000–2014 (Figure 5.10). Jackson (2007) noted an increase in Central Valley steelhead fishing effort prior to 2000 that was accompanied by a decrease in fishing effort observed on many coastal streams, and suspected that this may be the result of regulations allowing retention of hatchery-origin steelhead in the Central Valley.

Since the 2010 assessment there have been changes in fishing regulations that could have effects on freshwater fishery impacts. In March 2010, the hatchery-origin (adipose fin-clipped) steelhead bag limit increased from 1 to 2 fish on the Sacramento and American rivers (the possession limit increased from 2 to 4 fish as well). In March 2015, the bag and possession limit on adipose fin clipped steelhead increased on the Feather River, matching the previous regulations change on the Sacramento and American rivers. Recent drought conditions have affected some steelhead fishing opportunities for this DPS. For example, the California Fish and Game Commission imposed an emergency fishery closure on the American River in February of 2014. The closure ended in April of that year.

[14] Harvest impacts section prepared by Michael O’Farrell

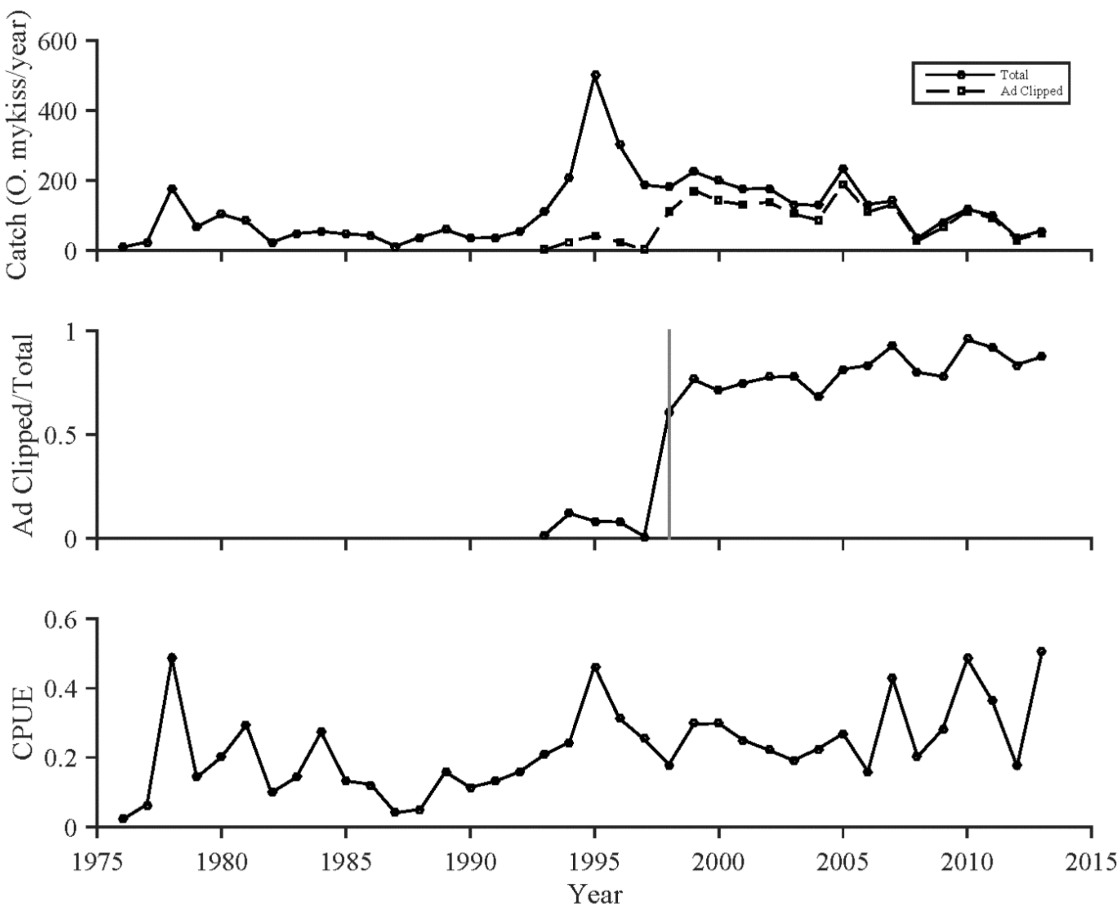

Figure 5.9. Top: Catch of steelhead at Chipps Island by the USFWS midwater trawl survey. Middle: Fraction of the catch with an adipose fin clip, 100% of hatchery steelhead have been marked (adipose fin clip) starting in 1998 (denoted with the vertical gray line). Bottom: Catch-per-unit-effort (CPUE) in fish per million m^3 swept volume. CPUE is not easily comparable across the entire period of record, as over time, sampling has occurred over more of the year and catches of juvenile steelhead are expected to be low outside of the primary migratory season.

California Department of Fish and Wildlife performs angler surveys on Central Valley streams, and data from these surveys are used to estimate steelhead harvest and fishing effort; however, these estimates do not appear to be regularly reported. No direct information is readily available on the level of CCV-Steelhead fishery impacts and an assessment of whether freshwater fishery impacts have increased in response to recent regulation changes cannot yet be made. Given this sparse information, it is likely that the level of impact has either not changed since the 2010 salmon and steelhead assessment (Williams et al. 2011), or has potentially increased due to increased bag and possession limits for hatchery-origin fish.

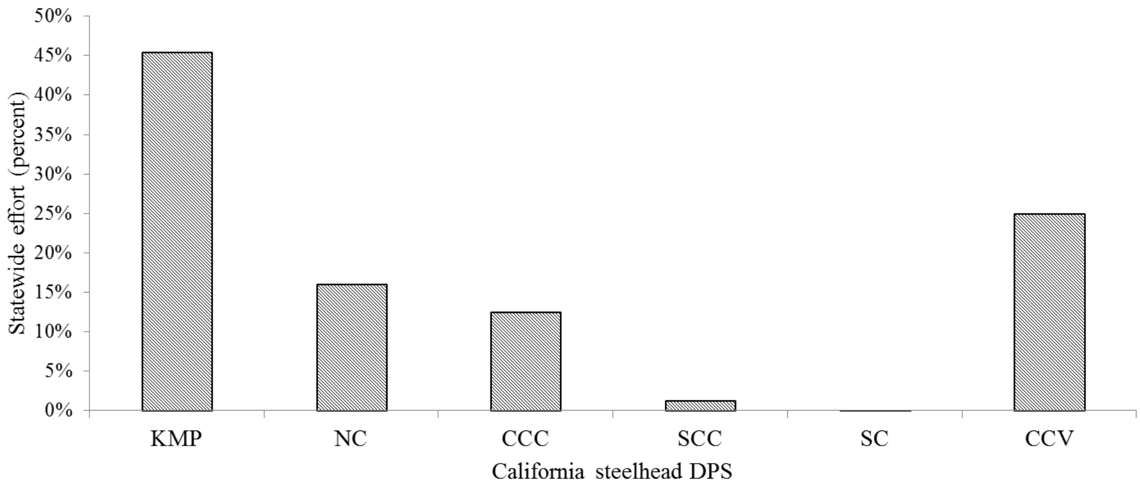

Figure 5.10. Distribution of California statewide steelhead fishing effort by DPS for years 2000−2014 (Jackson 2007; Farhat in preparation).

Summary and Conclusions

One of the greatest challenges in managing for resilient steelhead populations in our regulated rivers lies in understanding how water project operations promote, maintain, or suppress the expression and survival of the anadromous life-history form of *O.mykiss*. It is clear that some river habitats support almost exclusively abundant, non-anadromous populations, while others support the expression of anadromy (Satterthwaite et al. 2010). In the San Joaquin River tributaries specifically, there is great uncertainty in the extent to which the production of anadromous juveniles from tributaries is low and/or whether mortality of juvenile steelhead is so high during outmigration so as to preclude higher numbers of returning adult steelhead. While research suggests that the non-anadromous form can give rise to the anadromous form, it is possible that this only occurs at low levels, and it is more common for the steelhead form to give rise to the non-anadromous form (Donohoe et al. 2008). More studies are needed to understand the extent to which genes associated with the heritable components of anadromy could be lost from populations with low steelhead numbers, thus placing them at a greater risk of extinction.

The viability of the CCV-Steelhead DPS appears to have slightly improved since the 2010 assessment, when it was concluded that the DPS was in danger of extinction. This modest improvement is driven by the increase in adult returns to hatcheries from their recent lows, but the state of naturally produced fish remains poor. Improvements to the total population sizes of the three previously evaluated steelhead populations (Battle Creek, CNFH, and FRH), does not warrant a downgrading of the ESU extinction risk. In fact, the lack of improved natural production as estimated by samples taken at Chipps Island, and low abundances coupled with large hatchery influence in the Southern Sierra Nevada Diversity group is cause for concern. As in the previous assessments (Good et al. 2005; Williams et al. 2011), the CCV-Steelhead DPS continues be at a high risk of extinction.

6 South-Central/Southern California Coast Recovery Domain

David A Boughton[15]

Listed Distinct Population Segments

The domain is inhabited by two Distinct Population Segments (DPSs) of steelhead. The South-Central California Coast DPS (listed as Threatened) inhabits coastal stream networks from the Pajaro River system in Monterey Bay south to, but not including, the Santa Maria River system in Santa Barbara County. The Southern California Coast DPS (listed as Endangered) inhabits coastal stream networks from the Santa Maria River system south to the U.S. border with Mexico. For convenience I refer to fish of both DPSs as "southern steelhead."

Freshwater-resident (non-anadromous) *O. mykiss*, commonly known as rainbow trout, also occur in the same geographic region, frequently co-occurring in the same river systems as southern steelhead. Clemento et al. (2009) found that southern rainbow trout above impassable dams and southern steelhead below dams tended to be closely related genetically, suggesting that each steelhead DPS is simply the anadromous component of a corresponding Evolutionarily Significant Unit (ESU; Waples 1991) comprising both anadromous and non-anadromous *O. mykiss*. Anadromous and/or non-anadromous forms of the species also occur in some basins south of the U.S. border, on the Baja California Peninsula (Ruiz-Capos and Pister 1995).

Listing History and Initiation of Recovery Effort

The first comprehensive status review of steelhead was conducted by Busby et al. (1996), who characterized Evolutionarily Significant Units (ESUs) using the conceptual framework of Waples (1991), and then assessed extinction risk of each ESU. The South-Central California Coast and Southern California Coast Steelhead ESUs were subsequently listed as Threatened and Endangered, respectively, by NMFS under the U.S. Endangered Species Act (ESA). The original listing characterized the southern range limit as the southern end of the Santa Monica Mountains just north of Los Angeles, but it was later determined to occur further south, at least as far as the Tijuana River system at the U.S.-Mexico border, and possibly further south in Baja California. The listings were

[15] This section authored by D. Boughton is dedicated to Pete Adams, retired NMFS biologist who started the Science Center along this path of scientific recovery planning for California anadromous salmonids. A number of important planning efforts have been completed since the last viability assessment, including Federal recovery plans for each DPS (National Marine Fisheries Service 2009b, 2012b), and the design of a comprehensive monitoring plan that will track the extinction risk of each DPS over the long term (Adams et al. 2011). The recovery plans and the monitoring plan formally constitute the initiation of a recovery effort, in which actions affecting the fish either positively or negatively can be placed within the context of criteria for a viable metapopulation of the species. This viability assessment is thus the first to use these plans as a forward-looking frame of reference for updating the risk status of each DPS.

also modified to include only the anadromous component of each ESU, which are composed of both anadromous and non-anadromous forms of *O. mykiss*. Good et al. (2005) updated the status of Pacific coast steelhead populations five years after the listings, and another update was conducted in 2010 (Williams et al. 2011) and is available on-line. None of these updates led to changes in status of either listed DPS.

Consistent with ESA statute, the listings triggered the preparation of recovery plans. The first phase of recovery planning focused on the synthesis of scientific and technical guidance for recovering the two DPSs, and was conducted by NMFS Southwest Fisheries Science Center. This phase of planning was based on available scientific information and a conceptual framework for viable salmonid populations (McElhany et al. 2000). Findings are described in a series of NMFS Technical Memoranda describing ESU structure (Boughton et al. 2006, Boughton and Goslin 2006), viability criteria (Boughton et al. 2007), research needs (Boughton 2010c), a conceptual framework for recovery (Boughton 2010a), and a plan for ongoing monitoring of risk status of each DPS (Adams et al. 2011).

The second phase focused on preparation of recovery plans that describe strategies and goals for recovering the DPSs. Since the last viability assessment, the NMFS West Coast Region and its partners have formally adopted recovery plans for both DPSs (NMFS 2009b, 2012b). The plans are based on the biological needs of the fish and provide a foundation for restoring each DPS and its constituent populations to levels at which they would no longer be considered at risk of extinction.

These "levels" are formally known as viability criteria, and the summary statistics used to assess each DPS are known as viability metrics (e.g., Figure 6.1). With the publication of recovery plans and a monitoring plan, the goal of status review updates now becomes an assessment of whether viability metrics for each DPS are moving toward or away from the viability criteria. Unfortunately, this simple process of reviewing status is hampered at the moment by two problems: 1) scientific uncertainty about the viability criteria themselves, and 2) incomplete data on viability metrics. To address (1), below I review new information relevant to the viability criteria. To address (2), I review the implementation thus far of the monitoring plan, known formally as the California Coastal Monitoring Plan[16], or California CMP.

New Information Relevant to Viability Criteria

Risk status is based on the concept of viability at two levels of organization: the overall DPS, and individual populations composing the ESU of which the DPS is part.

[16] For information on the California Coastal Monitoring Program: http://www.calfish.org/ProgramsData/ConservationandManagement/CaliforniaCoastalMonitoring.aspx

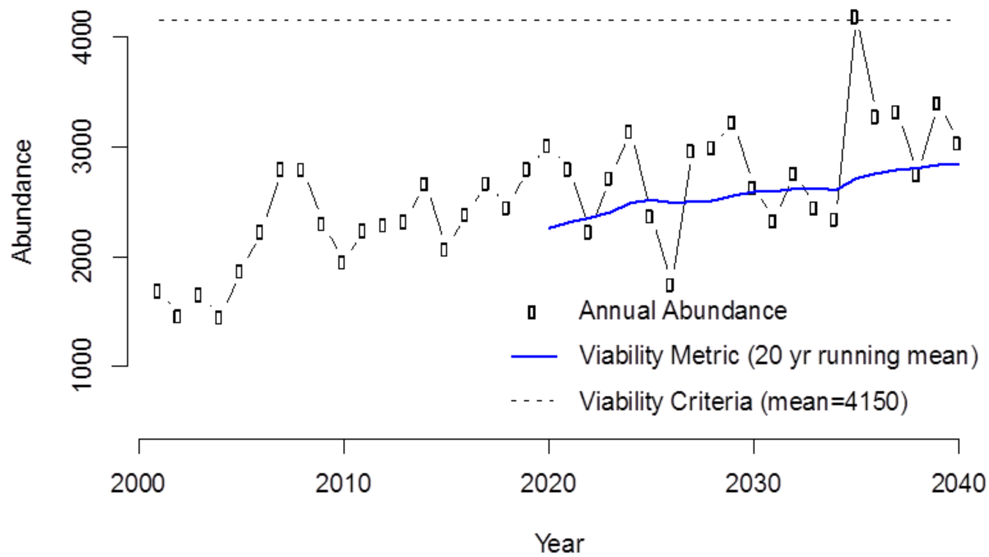

Figure 6.1. Concept of viability metric and a viability criterion applied to a hypothetical population.

DPS Viability

For each DPS, the recovery plans (NMFS 2009b, 2012b) developed viability criteria for populations (Table 6.1) and followed scientific recommendations by specifying a set of core populations on which to focus the recovery effort ("Core 1" and "Core 2" populations, Table 6.2). Formally, if each of these core populations were restored to viability (Table 6.1, top), and they also meet DPS-level criteria (Table 6.1, bottom), the DPS as a whole would be considered viable from a scientific perspective. However, there appear to be two discrepancies between the scientific recommendations for DPS viability (Table 6.1) and the list of core populations (Table 6.2):

First, scientific recommendations were that three populations in the Mojave Rim biogeographic area be restored to viability, but the recovery plan prioritizes only two (San Gabriel River, Santa Ana River) as either Core 1 or 2 populations. In addition, scientific recommendations were that eight populations in the Santa Catalina Gulf Coast area be restored to viability, but the recovery plan prioritizes only six (San Juan Creek, San Mateo Creek, San Onofre Creek, Santa Margarita River, San Luis Rey River and San Dieguito River) as either Core 1 or 2 populations. In the Recovery Plan four populations (San Diego, Sweetwater, Otay, and Tijuana) were designated as Core 1 or 2 populations, though Core 3 populations are recognized as important in promoting connectivity between populations, and genetic diversity across the DPS, and are therefore an integral part of the overall recovery strategy of the Recovery Plan. This approach is broadly consistent with the recommendations in the viability report, which noted that it is not clear if historically, the anadromous life history was consistently expressed in these populations of the extreme southern range limit.

Significant new genetic information bears on the question of native steelhead populations toward the southern extent of their range in California. Jacobson et al. (2014) analyzed genetic composition of *O. mykiss* sampled from a variety of sites in the Monte Arido, Mojave Rim and Santa Catalina Gulf Coast biogeographic areas (see also Abadia-Cardoso et al. in press). The majority of sites were found to harbor *O. mykiss* lineages

Table 6.1. Viability criteria emphasized in scientific recommendations.

Criteria for population viability

Prescriptive criteria:

Viability metric	Viability criterion	Notes
Mean annual run size	S > 4,150	Precautionary
Ocean conditions	Size criterion met during poor ocean conditions	
Population density	Unknown	Research needed
Anadromous fraction	100% of 4,150	Precautionary

Performance-based criteria:
One or more prescriptive criteria (above) could be replaced by a quantitative risk assessment satisfying the following:

1. Extinction risk of anadromous population less than 5% in the next 100 years.
2. Addresses each risk that is addressed by the prescriptive criteria it replaces.
3. Parameters are either a) estimated from data or b) precautionary.
4. Quantitative methods are accepted practice in risk assessment/population viability analysis.
5. Pass independent scientific review.

Criteria for DPS viability

Viability metric	Viability criterion
Biogeographic diversity	1. Sufficient numbers of viable populations in each biogeographic group (See Table 6 in Boughton et al. 2007).
	2. Viable populations inhabit watersheds with drought refugia.
	3. Viable populations in basins separated by > 68 km if possible.
Life-history diversity	Viable populations exhibit three life-history types (fluvial-anadromous, lagoon-anadromous, resident)

Table 6.2. Core populations designated by recovery plans for recovering to viability.

Population	Adult abundance[a]	Spatial structure[a]	Smolt counts[a]
<u>South-Central California Coast DPS</u>			
Interior Coast			
Pajaro River	N	I	N
Salinas River	Y	I	B
Carmel River			
Carmel River	B	I	N
Big Sur Coast			
San Jose Creek	N	N	N
Little Sur River	N	N	N
Big Sur River	B*	N	N
San Luis Obispo Terrace			
San Simeon Creek	N	N	N
Santa Rosa Creek	N	N	N
San Luis Obispo Creek	B*	N	N
Pismo Creek	N	N	N
Arroyo Grande Creek	N	N	N
<u>Southern California Coast DPS</u>			
Monte Arido Highlands			
Santa Maria River	N	N	N
Santa Ynez River	B	Y	B
Ventura River	B	Y(I)	B
Santa Clara River	B	N	B
Santa Barbara Coast			
Canada de la Gaviota	N	N	N
Goleta Slough complex	N	N	N
Mission Creek	N	N	N
Carpenteria Creek	Y	N	N
Rincon Creek	N	N	N
Santa Monica Mountains			
Arroyo Sequit	B*	Y	Y
Malibu Creek	B*	Y	Y
Topanga Canyon	B*	Y	Y

Table 6.2. continued.

Population	Adult abundance[a]	Spatial structure[a]	Smolt counts[a]
Mojave Rim			
San Gabriel River	N	N	N
Santa Ana River	N	N	N
Santa Catalina Gulf Coast populations			
San Juan Creek	N	N	N
San Mateo Creek	N	N	N
San Onofre Creek	N	N	N
Santa Margarita River	N	N	N
San Luis Rey River	N	N	N
San Dieguito River	N	N	N

a – Y = yes, N = no, B = estimates are likely biased (B* = redd counts, which can be bias-corrected with data from life-cycle monitoring stations), I = index reaches rather than randomly sampled reaches or complete census of anadromous habitat.

derived from hatchery stocks of rainbow trout rather than native coastal steelhead lineages. Native lineages were generally found throughout the Monte Arido sites, but most of the Mojave Rim and Santa Catalina Gulf Coast sites consisted of non-native hatchery lineages, “representing almost complete introgression or replacement of native fish by introduced hatchery rainbow trout” (Jacobson et al. 2014). Three groups of sites contained significant evidence of native steelhead ancestry: 1) the San Luis Rey River population, 2) Coldwater Canyon, tributary to the Santa Ana River, and 3) the San Gabriel River population, except for sites on the Iron Fork and Devil’s Canyon Creek that showed hatchery lineage. These three groups of sites are part of three core populations listed in Table 6.2. A few other sites, especially Bear Creek, tributary to the Santa Ana River, and Devil’s Canyon Creek, tributary to the San Gabriel River, showed detectable signals of native ancestry co-existing with a strong signal of hatchery lineages. The authors of the report concluded that “overall, relatively few populations [sites] in this study appear to be pure native southern California *O. mykiss*” (Jacobson et al. 2014), but they also noted that some of the non-native genetic introgression may increase the potential for evolutionary adaptation to changing conditions and might therefore contribute to viability.

The second discrepancy is that scientific recommendations emphasized that core populations be situated in watersheds with drought refugia (Table 6.1, bottom). There does not appear to be any systematic information on the distribution of drought refugia, even though the current drought provides a valuable opportunity to identify such refugia. Thus it is unclear if the selected set of core populations meets this criterion.

Given the current drought, it might be useful to quote from the viability report: "…tree-ring data described by Cook et al. (2004) go back to the year 800 A.D., and record at least four multi-decade droughts prior to 1300 A.D. These events had far greater magnitudes than anything observed during the historical period. The aboriginal steelhead populations must have either survived in drought-resilient refugia, or have been regionally extirpated prior to 1300 A.D. and recolonized in the subsequent centuries. If the refugium hypothesis is correct, ESU viability is probably contingent on forecasting the location of refugia under future climate regimes. If the recolonization hypothesis is correct, ESU boundaries are currently misspecified. Evaluation of the refugium hypothesis, particularly as it relates to future climate, is an obvious research priority." (Boughton et al. 2007).

Population Viability

Viability criteria at the population level are summarized in the top of Table 6.1. In the scientific recommendations (Boughton et al. 2007), there was broad agreement that the viability metrics of Table 6.1 were sufficient for assessing risk, but also agreement that the specific viability criteria were highly sensitive to scientific uncertainty about key aspects of steelhead ecology. These key knowledge gaps included 1) uncertainty about the magnitude of normal fluctuations in adult abundance, and 2) uncertainty about the underlying biological mechanisms for expression of life-history diversity, especially factors triggering anadromous versus non-anadromous life-histories within populations. Thus the criteria that mean annual spawner abundance 1) be greater than 4150, and 2) be composed of 100% anadromous fish, were recommended as a risk-averse approach. It was expected that further scientific work would either support these criteria or allow one or both to be relaxed, depending on results.

The last five years have seen little progress in developing better scientific information on population fluctuations, but significant progress on maintenance of life-history diversity. However, there has been no work on how the ecological and biological factors that maintain life-history diversity at the population level bear on the viability criterion for anadromous fraction.

Data on population fluctuations will emerge over time with the implementation of the CMP, discussed further in the next section. The CMP emphasizes annual estimates of abundance of anadromous adults in each Core 1 and Core 2 population, which is intended to provide data on abundance and productivity metrics, including abundance fluctuations. Missing from the CMP but just as important with respect to future revision of viability criteria, are ongoing monitoring of abundance and fluctuations of the non-anadromous life-history type in each population over time, and also the lagoon-anadromous form (Boughton et al. 2007).

Maintenance of Life-history Diversity

Previous research led by NMFS and UC Santa Cruz suggested that diversity of life histories (anadromous versus non-anadromous, age of smolting and age of maturation) was largely controlled by diversity in growth rates during the early life history of the fish (Bond et al. 2008, Satterthwaite et al. 2009; Beakes et al. 2010; Satterthwaite et al. 2012), and thus was largely under ecological control. On the other hand, numerous studies have demonstrated the heritability and genetic influence on expression of anadromy (Kendall et al. 2015). In particular, a recent analysis identified an important genetic component on chromosome *Omy5* (Martinez et al. 2011; Pearse et al. 2014; Pearse et al. in preparation). Evidently, a portion of *O. mykiss* chromosome 5 has undergone an inversion, in which a segment of the chromosome has been reversed end to end in some fish but not others. This inversion is passed on to progeny, but for fish in which one chromosome is inverted and the other not (i.e., a parent of each type), no crossing-over can occur during meiosis, and so the set of genes on the inverted section of chromosome are tightly linked (prevented from mixing between the two chromosome types). Such tightly linked sets of genes are sometimes called "supergenes."

Pearse et al. (2014) surveyed the occurrence of these two chromosome types in existing genetic samples from throughout the California coastal mountains, and found several interesting patterns:

1) Both chromosome types were present at most sites,
2) There was strong evidence of selection on the set of linked genes within the inversion,
3) One chromosome type dominated sites in anadromous waters, whereas the other chromosome type dominated sites in formerly anadromous waters that are now upstream of impassable dams.

Pearse et al. (2014) concluded that natural selection favors one chromosome type in anadromous waters, and this chromosome type therefore likely plays a role in maintaining the anadromous life-history, and natural selection favors the other chromosome type in non-anadromous waters, and therefore it likely plays a role in maintaining the non-anadromous life history. However, both chromosome types do occur in both types of waters, and both life-histories are observed in anadromous waters, so the relationship is probably not a simple association between non-anadromous and anadromous genomic elements.

Pearse et al. (In preparation) combined genetic analysis of the Omy5 inversion with a mark-recapture study of juvenile *O. mykiss* in a small population in the Big Sur biogeographic group. For age 0 fish, the probability of emigrating from freshwater to the ocean was associated with chromosome type, sex, and juvenile body size, and also interaction effects for these three traits. However, the associations were probabilistic rather than "complete": emigrants included juveniles of both sexes, a broad range of sizes (100 – 250 mm), and both chromosome types. Pearse et al. (In preparation) conclude that the Omy5 inversion region represents a "supergene with a major effect on a complex behavioral trait (i.e., migration)", but that the individual component genes have not yet been resolved, and also that chromosome Omy12 "also contains regions important for smoltification-related traits… In addition, other genomic regions, heritable epigenetic effects, and subtle population structure or assortative mating may also affect this complex life-history trait." Rundio et al. (2012) also described evidence that females were more likely than males to emigrate in this study population, and Ohms et al. (2014) documented similar female-biased emigration in nine populations distributed broadly across the Pacific Northwest, southern Alaska, and northern California.

These new findings demonstrate that non-anadromous and anadromous life-histories in *O. mykiss* in the southern domain and elsewhere are tightly integrated. This suggests that the viability criterion for a 100% anadromous fraction in core populations (Table 6.1) should be revised. However, the studies summarized above do not include any population-viability analyses, which would be necessary for proposing a specific revision of the criterion.

New Information on Methodology for Viability Metrics

The CMP draws on the VSP framework of McElhany et al. (2000) to assess viability in terms of four population metrics: abundance, productivity, spatial structure, and diversity. The CMP also outlines the creation of a system of Life-Cycle Monitoring stations (LCMs) to collect additional data necessary for the interpretation of those four metrics (Adams et al. 2011). The CMP is intended to provide data sufficient to conduct viability assessments and status reviews under the U.S. Endangered Species Act, but at present is only partially implemented. Here I review methodological issues that appear to be impeding implementation; in the next section I review the level of implementation thus far.

According to Adams et al. (2011), the CMP divides the coastal zone of California into northern and southern areas based on differences in species composition, levels of abundance, distribution patterns, an d habitat differences that require distinct monitoring approaches. The South-Central California Coast and Southern California Coast Steelhead DPSs are in the southern area. Implementation of the CMP in the southern area means monitoring the following metrics in the core populations listed in Table 6.2 (Adams et al. 2011):

1) Unbiased estimates of annual anadromous run size, for tracking abundance and productivity.
2) Unbiased estimates of the spatial distribution of juveniles, possibly also in lower priority populations, for tracking spatial structure.
3) Unbiased estimates of annual smolt production in a subset of Table 6.2 populations that are well-distributed biogeographically (life-cycle monitoring stations), for distinguishing between changes in ocean conditions and freshwater conditions.
4) Unbiased estimates of diversity metrics, still to be determined, for tracking diversity.

Here, “unbiased” is used in the statistical sense of estimators whose long-run sampling distribution is equal to the parameter being estimated—for example, methods that do not systematically undercount or overcount fish over repeated surveys. Below I summarize methodological progress on estimating these four metrics.

Abundance and Productivity

In both northern and southern monitoring areas, the assessment of abundance and productivity is based upon unbiased estimates of the annual number of anadromous adults across each ESU, with productivity calculated as the trend in anadromous adults over time. In the northern area (Santa Cruz area north to Oregon), adult abundance is estimated via redd surveys conducted in a spatially balanced, stratified-random sample of stream reaches, and bias-corrected by redds-per-female estimates obtained from LCMs. At the time of CMP development, redd surveys were believed to be infeasible in the southern area due to the extremely episodic flow regime and high bed loads (movement of sand and gravel) during the spawning season, as well as inaccessibility of many upland tributaries during the rainy season. Instead the CMP specified that abundance be estimated by counting upstream migrants at fixed counting stations in the lower mainstems of rivers, but was somewhat agnostic about how it would be done.

To fully support a viability assessment such as this one, such counting would need to occur in the full complement of populations listed in Table 6.2. However, counting would not necessarily need to occur in every population in every year; a rotating-panel sampling plan could probably be used, similar to the sampling of reaches used for redd surveys in the northern area, but with sampling units being whole populations rather than individual stream reaches. That is, some of the populations in Table 6.2 would be counted every year, others would be counted every 3 or 4 or 12 years on a staggered schedule. This is not something envisioned in the original CMP, but would be consistent with its goals and more efficient to implement.

Since the development of the CMP strategy outlined in Adams et al. (2011), there appear to have been two efforts to conduct redd surveys in the southern area, with mixed results. The Monterey Peninsula Water Management District has conducted redd surveys in the lower Carmel River as District resources have permitted, but could not fully implement the protocols used in the northern area (e.g., Gallagher and Gallagher 2005). These protocols specify that sampled reaches be surveyed every two weeks for the duration of the spawning season, which was not possible in the lower Carmel River due to high flows associated with the episodic flow regime, probably leading to an undercount of redds (K. Urquhart, MPWMD, personal communication). Alternately, the NMFS West Coast Region office in Long Beach has had success conducting redd surveys in the Ventura River that adhere closely to the northern area protocol, though these data have not been continued for sufficiently long enough to support a viability assessment (R. Bush, NMFS, personal communication).

These efforts suggest that redd surveys might be able to produce unbiased estimates of adult abundance in some situations but not others. In situations where they appear feasible, such as the Ventura River system, redd surveys would need to be bias-corrected using estimates of redds-per-female estimated at life-cycle monitoring stations (Adams et al. 2011). If redd surveys were to become a strategy for implementing the CMP in the southern area, they would probably not be a universal solution as in the north. The problem with sampling during high flows is also encountered in the northern area (D. McCanne, CDFW, personal communication). The problem with sampling in inaccessible mountain tributaries during the rainy season has not yet been addressed.

At the time of CMP development, one of the most promising methods for counting anadromous adults was the new DIDSON acoustic camera (Pipal et al. 2010a; Pipal et al. 2010b; Pipal et al. 2012). These have started to be deployed in the domain, currently in the Carmel River, Ventura River, Carpenteria Creek, and Salsipuedes Creek (tributary of the Santa Ynez River). There appear to be three problematic methodological issues. The most important is that in some situations, migrating steelhead go back and forth a lot ("milling"), so that the counts of adult steelhead are really the net difference between upstream migrants and downstream migrants. If significant numbers of adult steelhead survive spawning, and migrate downstream to the ocean as kelts, then kelts and "millers" would be confounded, leading to biased estimates. Two other methodological issues are species identification and the sheer number of person-hours required to review DIDSON output in order to produce the counts. The latter issue should be amenable to improvement by using machine-learning techniques to aid in image interpretation. This is a promising avenue for research that might lead to cheaper, more efficient DIDSON monitoring.

Various other methods have been or are starting to be used to count anadromous adults, such as monthly snorkel surveys in Topanga Creek (Stillwater-Sciences et al. 2010), trapping stations in tributaries of the Santa Ynez River (Robinson et al. 2009), a visual imaging system at a fish passage facility on the Salinas River (Cuthbert et al. 2014a), and a counter on a fish ladder on the Carmel River (MPWMD 2013). In addition, a method has been proposed to use two-stage sampling and PIT-tagging of juveniles combined with monitoring of migrants (Boughton 2010b). I summarize data from these sources and methodological issues later in this section, in the update on the viability of Distinct Population Segments. The most important methodological issues appear to be 1) the need to consistently provide unbiased estimates of adult abundance, for example by estimating observation or capture probabilities and by use of randomly sampled stream reaches rather than subjectively chosen index reaches; and 2) the need for methods suitable for the normal range of environmental conditions expected for the domain, which typically involve extreme flow events, high bedloads, and remote rivers and tributaries that are difficult to access during the wet season.

Spatial Structure

The CMP recommends that spatial structure be monitored using summer and fall snorkel surveys that count juveniles in a stratified-random, spatially balanced sample of reaches (Adams et al. 2011). The sampling is achieved using Generalized Random Tesselation Stratified (GRTS) sampling to achieve spatial balance, and a rotating panel design to achieve a balance between the need to estimate structure at a particular time, and the need to estimate trends in structure over time. This is the same sampling framework used in the northern CMP area for both redd surveys and juvenile surveys.

To my knowledge, no such data have been collected in either DPS in the last five years. Topanga Canyon and Santa Ynez River have received comprehensive snorkel surveys, the former for over a decade (Stillwater-Sciences et al. 2010), but no broad-scale data using reach-sampling have been produced. California Department of Fish and Wildlife is in the process of developing a ground-truthed sampling frame for the Santa Barbara

Coast (D. McCanne, CDFW, personal communication) and for Monterey County (J. Nelson, CDFW, personal communication).

Diversity

At the time of CMP development, diversity traits were not sufficiently understood for their monitoring to be specified. Adams et al. (2011) stated that "local diversity traits will need to be surveyed, eventually leading to local diversity monitoring plans. Specific projects targeting both broad and focused levels and patterns of genetic diversity will be developed. Tissue collections for these projects will be coordinated with other CMP activities." We are now in a better position to propose some diversity traits that need to be monitored to assess viability. The viability criteria (Table 6.1, see also Boughton et al. (2007)) emphasize the critical importance of resident adults. The findings of Pearse et al. (2014) and Jacobson et al. (2014) show the importance of genetic information for assessing viability, both in terms of genetic heritage (e.g., native vs. hatchery introductions) and in terms of occurrence of the supergene variants.

Diversity metrics in the form of unbiased estimates of non-anadromous adults and the distribution and diversity of genetic polymorphisms, could all be integrated in a straightforward manner with the broad-scale juvenile sampling that the CMP specifies for spatial structure. An important methodological change would be required: Collection of genetic samples requires handling the fish, which means that mark-recapture or depletion electrofishing would need to occur at a subsample of the reaches selected for juvenile snorkel counts. Such subsampling would also allow the snorkel counts to be bias-corrected (Boughton et al. 2009). If methods were developed to distinguish juveniles from non-anadromous adults in both snorkel counts and electrofishing samples, an unbiased estimate could then be made of the number of non-anadromous adults in the sampling domain. Additionally, tissues could be taken from electrofishing sites for genetic analysis that would provide unbiased estimates of various gene frequencies. I recommend that updates to the CMP be considered that include such diversity monitoring.

Environmental DNA might provide another avenue for monitoring genetic diversity, but its statistical properties for inferring unbiased gene frequencies in steelhead populations is not clear.

Life-Cycle Monitoring Stations

According to Adams et al. (2011), LCMs are a fundamental component of the CMP that delivers two functions: providing unbiased estimates of ocean survival so that changes in salmonid numbers can be parsed into changes due to freshwater versus marine conditions; and as "magnets for other kinds of recovery-oriented research, particularly studies of fish habitat-productivity relationships and evaluations of habitat restoration effectiveness." For the first function (estimating marine survival), an LCM needs three attributes: 1) annual, unbiased estimates of anadromous adults, 2) annual, unbiased estimates of smolt production, and 3) a sufficiently large number of anadromous adults to provide accurate estimates of marine survival (at least 20 per year, preferably more than 100 anadromous adults each year).

Methodological issues for estimating anadromous adults were described above in the section on abundance and productivity.

Methodological issues for estimating smolt production have seen little progress since the last assessment (Williams et al. 2011) and remain problematic. Originally the DIDSON acoustic camera seemed promising as a tool for estimating smolt production, but the size of smolts is close enough to the resolution of DIDSON imagery that detection probability is probably substantially less than 1 (K. Pipal, UCSC/NMFS SWFSC, personal communication). Fyke nets, traps, and visual imagery at fish passage facilities, developed for counting anadromous adults, are also being used to count smolts, but with qualified success. The main problem is counts that are likely biased low due to failure of counting stations during high flow events. Two other problems are distinguishing smolts from juvenile downstream migrants (typically age-0 or age-1 fish moving down to the estuary near the end of smolting season and in early summer), and the difficulty of estimating smolt body sizes. Although estimates of smolt body sizes were not emphasized in the CMP, we should expect marine survival to involve strong interaction effects between ocean condition and smolt size at ocean entry (Ward 2000, Bond 2006). If this were not accounted for then some unknown component of change in marine survival may instead be due to changes in freshwater condition via its effect on smolt body size.

Boughton (2010b) described a framework for using PIT tags to estimate both smolt production and adult abundance. PIT tags would be implanted in juveniles collected from reaches sampled from a stream network, and thus would be straightforward to integrate with the reach-sampling methods used for spatial structure (described earlier). Smolt production is estimated from the proportion of tagged fish that are detected at a downstream tag-reading station near the mouth of the river. An application of this approach in the southern domain has not yet been described, but some advantages and disadvantages are already clear. Advantages are that the method could be integrated with spatial-structure sampling; could provide information on smolt size (via pre-smolt size at the time of sampling); and since the originating reaches of tagged smolts would be known, it could provide a powerful tool for evaluating habitat-productivity relationships, including testing of various habitat-restoration actions, regulatory actions, or flow-management actions relative to “control” reaches. Disadvantages are that progress is still needed for designing reader stations (particularly antennae) that are robust to high-flow events, and that over time this approach is likely to lead to an accumulation of tags in the river bed (from dead juveniles) (D. Rundio, NMFS SWFSC, personal communication). These “ghost or rogue tags” get moved by high flow events and cannot be readily distinguished from live smolts, thus generating overestimates of smolt production. The bias would tend to increase over time as tags accumulate, such that the ghost tags would generate a “ghost recovery” of smolt production.

Monitoring of Viability Metrics

Below I summarize viability metrics that are currently being collected in the domain. In general the metrics are not formally assessed because the period of record is too short for such assessment to be meaningful.

Interior Coast Range

No adult counts or smolt counts have been made in the Pajaro River. The organization Coastal Habitat Education and Environmental Restoration has rescued a mean of 12 adult steelhead per year (sd = 20) from 2006 through 2013 (J. Casagrande, NMFS West Coast Region, personal communication), suggesting consistent occurrence of at least modest numbers of anadromous fish. Some limited assessments of spatial structure have been made in Uvas Creek, Llagas Creek, and Corralitos Creek since 2005, using backpack electrofishing at index reaches (Casagrande 2014), but there do not appear to be unbiased estimates of spatial structure based on stratified-random sampling.

The Salinas River has an established counting station in operation since 2011 (Cuthbert et al. 2014a), with a mean of 22 (sd = 22) total upstream migrants per year. Also reported are net upstream migrants (total upstream migrants minus total downstream migrants) with a mean of 18 (sd = 18) migrants per year. Smolt production has also been monitored with rotary screw traps since 2010, but the counts are likely biased low due to incomplete coverage of the migration season and low (unquantified) trap efficiency during some flow conditions (Cuthbert et al. 2014b). Juvenile abundance has been estimated via backpack electrofishing at eight index reaches since 2010 (Monterey County Water Resources Agency 2014). There do not appear to be unbiased estimates of spatial structure based on stratified-random sampling.

Carmel River

The Carmel River is the only population within the domain for which there is a time-series of adult abundance longer than 20 years. Unfortunately the counts probably have a bias that has changed over time, because the counting has occurred at San Clemente Dam and misses adults that spawn in the river downstream of the dam. This downstream area has been an area of extensive habitat restoration in the past 15 years, so the number of fish spawning here has likely increased and thus the negative bias in the counts has probably also increased over time (K. Urquhart, MPWMD, personal communication).

A plot of the counts (Figure 6.2) shows interesting variation over time. A period of zero counts from 1988 to 1991 were due to a drought, during which local water users drew down the water table and the lower river remained continuously dry and offered no opportunities for migration. During this period the Carmel River Steelhead Association used a nearby seawater facility to operate a broodstock program, releasing many mature anadromous adults as well as hundreds of thousands of juveniles to the river system (Thomas 1996). Numbers quickly climbed after the end of the drought (and the broodstock program) in 1991.

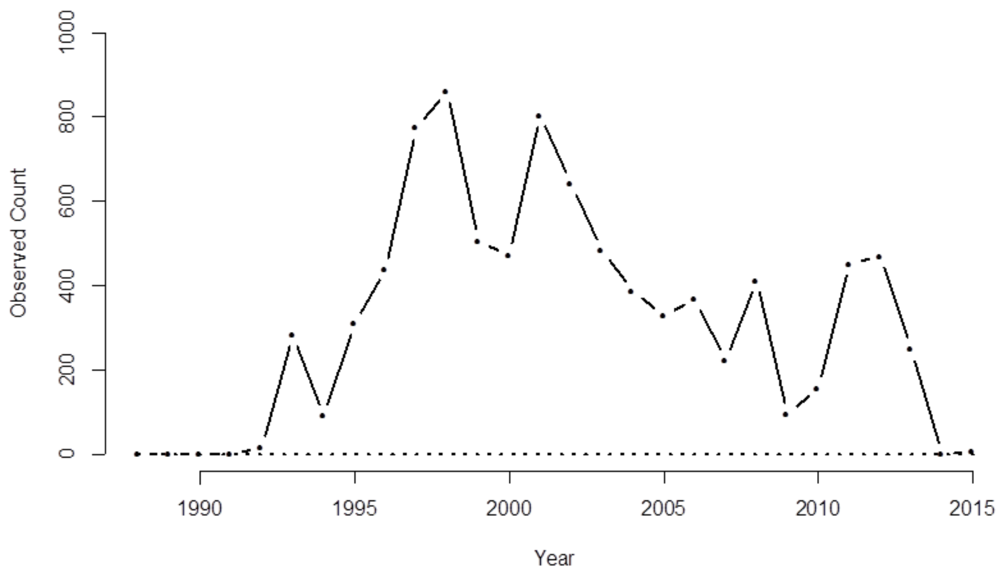

Figure 6.2. Adult steelhead counted at San Clemente Dam on the Carmel River since 1988.

The past 20 years (1996–2015) has seen a consistent though irregular decline in numbers (Figure 6.2), with an average decline of 16.5% per year (or about 50% per generation, assuming a 4-year generation time). Low counts in 2014–2015 are almost certainly due to drought, but the decline was clearly underway prior to 2014.

The 20-year decline coincides with a period of intense management aimed at recovering steelhead, including a restoration of estuary habitat, restoration of riparian vegetation, partial restoration of water tables, and a captive-rearing program for juveniles that get stranded in drying sections of river during the summer. One possible explanation for the decline is that improved conditions in the lower river motivate many adults to stop and spawn prior to reaching the dam and getting counted. However, the local water district (Monterey Peninsula Water Management District, MPWMD) has conducted occasional redd surveys, and found that the number of redds downstream of the dam do not fully account for the decline (see previous viability assessment, Williams et al. 2011), supporting that there has been a decline in abundance.

(Arriaza in review) describes the application of a life-cycle model to steelhead data in the Carmel River. The analysis suggests the decline is due to a long-term decline in the growth rates of age-0 juveniles in the river, which reduces the smolting rate and the survival of smolts once they enter the ocean. The decline in growth rates has apparently led to a switch from most anadromous adults being the result of in-river wild production, to most anadromous adults being fish released from the Sleepy Hollow Steelhead Rearing Facility. From 2005 onward, the vast majority of production of anadromous adults appears to have come from the rearing facility (Arriaza in review, see Figure 2.19).

A notable restoration event in the Carmel River during the past five years has been the removal of San Clemente Dam, and the rerouting of the river channel around the large stockpile of sediment that had accumulated upstream of the dam during the prior 90 years. Dam removal and the completion of the re-route channel have been completed as of December 2015, in time for the 2016 water year which commences 1 Oct 2015. The ecological effects of the dam removal on downstream habitats and on steelhead population viability will provide valuable information for future restoration efforts.

Big Sur Coast

National Marine Fisheries Service Southwest Fisheries Science Center has conducted a tagging study of steelhead in Big Creek (Core 3) since 2004, but has not used it to estimate abundance of anadromous adults, spatial structure of juveniles, or smolt production.

California Department of Fish and Wildlife has conducted redd surveys in the Big Sur River in 2012 (first field test) and in the 2014 and 2015 spawning seasons (J. Nelson, CDFW, personal communication). Each year they surveyed the entire anadromous portion of the stream network, using the field protocols established by Gallagher and Gallagher (2005) (T. Anderson, CDFW, personal communication). Snorkel surveys of juveniles were conducted in 2011 to provide a snapshot of spatial distribution, but have not been continued. Investigation is underway for installation of a DIDSON monitoring site, data from which would provide a basis for estimating redds per female, one of the functions of a LCM station.

There is no apparent monitoring of viability metrics in San Jose Creek and Little Sur River, the other Core 1/Core 2 populations in this biogeographic group.

San Luis Obispo Terrace

The city of San Luis Obispo initiated redd surveys in 2015, and plans to continue the effort using field protocols developed in the Ventura River by NMFS West Coast Region - Long Beach Office (F. Otte, NMFS, personal communication). There is no apparent monitoring of viability metrics in the four other Core 1/Core 2 populations in this biogeographic group.

Monte Arido Highlands

The Santa Maria River population does not appear to be monitored for any of the viability metrics. In the Santa Ynez River, adult and smolt counts have been collected since 2001 via migrant trapping (Cachuma Operation and Maintenance Board 2013), but the counts are likely biased low due to inability to trap during high flows and focus of trapping effort on two key tributaries rather than the whole river system (Robinson et al. 2009). From 2001 to 2011 (the latest date for which counts are published), the mean number of anadromous adults trapped per year was 3.4 (sd=5.2) and the mean number of smolts trapped per year was 146 (sd=116). California Department of Fish and Wildlife initiated DIDSON counts in a tributary (Salsipuedes Creek) in 2013 but has not yet released a report. Comprehensive snorkel surveys have been conducted since 2001 by Cachuma Operation and Maintenance Board, and may be suitable for estimating spatial structure if evaluated at the reach level.

In the Ventura River, the Casitas Municipal Water District (CMWD) issues annual reports on movement of *O. mykiss* through the Robles Fish Passage Facility. The most recent report was 2013 (CMWD 2013). Currently, counts do not distinguish adult steelhead or smolts from other age classes of the fish. Allen (2014) surveyed spatial structure from 2006 to 2012 using a combination of snorkel surveys and electrofishing of juveniles. Rather than using GRTS sampling, Allen (2014) used a three-stage hierarchical sampling scheme in which the first stage was subbasin, the second stage used index reaches, and the third stage used random selection of sites within index reaches.

For the Santa Clara River, the United Water Conservation District issues annual reports describing counts of adult steelhead and smolts passing through the Freeman Diversion Facility in the lower river. The most recent report was 2013 (Howard and Booth 2013), when zero (0) anadromous *O. mykiss* and zero (0) non-anadromous *O. mykiss* were observed. In general these counts represent lower bounds on abundance, as they do not enumerate fish that pass over the low diversion dam itself.

Santa Barbara Coast

California Department of Fish and Wildlife initiated DIDSON counts in Carpenteria Creek in 2014; data are not yet available (D. McCanne, CDFW, personal communication). California Department of Fish and Wildlife is developing a sampling frame and plans to initiate spatial-structure sampling in other populations of the biogeographic group. They have conducted pilot surveys in Gaviota Creek, Refugia Creek, and Arroyo Hondo.

Santa Monica Mountains

The core 1 and 2 populations in the Santa Monica Mountains are Arroyo Sequit, Malibu Creek, and Topanga Creek, and population data for each are being collected by the Resource Conservation District of the Santa Monica Mountains (Dagit et al. 2015). Snorkel surveys have been conducted monthly in reaches of each creek “where the majority of *O. mykiss* were confined due to either low water levels…or in Malibu below Rindge Dam” (Dagit et al. 2015). A random sample of reaches had multi-pass dives to calibrate detection probabilities. Life stages were visually classified using a rating protocol. “Smolt” counts (scare quotes as in the original report) were generated from the snorkel data using the visual classification. Redd counts were also made during the snorkel surveys (i.e., once per month), and twice per month since 2011 in Topanga Creek during the January – May spawning season.

Mojave Rim

No apparent monitoring of viability metrics.

Santa Catalina Gulf Coast

No apparent monitoring of viability metrics.

Harvest Impacts[17]

South Central California Coast Steelhead − Ocean harvest of steelhead is extremely rare, and is in particular an insignificant source of mortality for South Central California Coast (SCCC) steelhead. While insufficient data exists to estimate SCCC steelhead freshwater exploitation rates, these rates are likely relatively low given California's prohibition of natural-origin steelhead retention. Fishing effort estimates based on angler self-report cards are available for 2000–2014 which suggest very low levels of effort for this DPS over this period (Figure 6.3). Beginning in 2013, fishing regulations for many streams changed from allowing no steelhead retention to allowing a daily bag limit of two hatchery-origin steelhead per day. In summary, while no direct information is available on the level of SCCC steelhead fishery impacts, it is reasonable to conclude that the level of impact has either not appreciably changed since the 2010 salmon and steelhead assessment (Williams et al. 2011), or potentially increased due to increased bag limits for hatchery-origin fish.

Southern California Coast Steelhead − Ocean harvest of steelhead is extremely rare, and is in particular an insignificant source of mortality for Southern California Coast (SCC) steelhead. While insufficient data exists to estimate SCC steelhead freshwater exploitation rates, these rates are likely relatively low given California's prohibition of natural-origin steelhead retention. Fishing effort estimates based on angler self-report cards are available for 2000–2014 which suggest extremely low levels of effort in this DPS over this period (Figure 6.3). While no direct information is available on the level of SCC steelhead fishery impacts, it is reasonable to conclude that the level of impact has not appreciably changed since the 2010 salmon and steelhead assessments (Williams et al. 2011).

Summary of Findings

• The prevalence of extensive non-native ancestry in Mojave Rim and Santa Catalina Gulf Coast shows that risk status of the Southern California Coast Steelhead DPS is greater than previously thought. Native lineages have been nearly extirpated from this far

[17] Harvest impacts section prepared by Michael O'Farrell

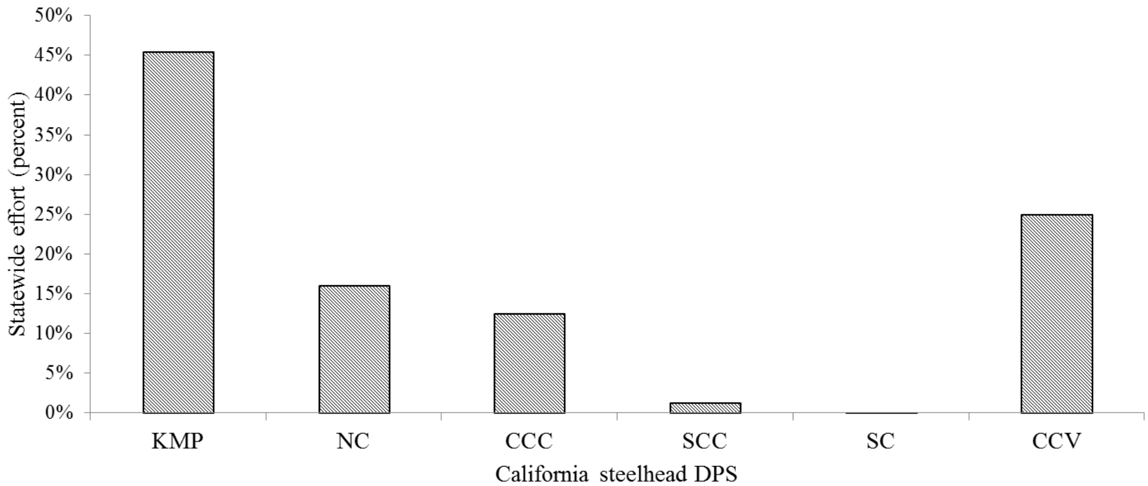

Figure 6.3. Distribution of California statewide steelhead fishing effort by DPS for years 2000–2014 (Jackson 2007; Farhat in preparation).

southern region of the native range of *O. mykiss*. Introduced lineages, primarily from the California Central Valley Steelhead DPS, are extant, introgressing with, and in some cases replacing native lineages. Presumably these introduced lineages have begun to evolutionarily adapt to the local habitats, but do not have the long history of adaptation that the native lineages had. Their potential role in the recovery of the species is not clear.

• There has been a fairly steady 15-year decline in abundance of anadromous adults in the Carmel River, the one population in the southern domain with a reasonably long history of monitoring. This decline is somewhat surprising since it coincides with a concerted effort to restore habitat in the river system and to improve numbers through a rescue/captive-rearing operation. The decline indicates an increase in extinction risk in the South Central California Coast Steelhead DPS, though it is likely that abundance in other populations show different patterns, and possible that such patterns would show that risk is holding steady or even improving (i.e., lower extinction risk).

• Currently, viability cannot be adequately assessed due to lack of implementation of the California Coastal Monitoring Plan (CMP). We recommend:

- Full implementation of CMP abundance monitoring and spatial-structure monitoring,
- Adding to the CMP the monitoring of non-anadromous adults and genetic diversity,
- Greater emphasis on monitoring methods that are unbiased or can be bias-corrected,
- Site-selection and initiation of additional Life-Cycle Monitoring stations. These could serve as study sites to clarify the role of the chromosome inversion in the

maintenance of life-history diversity, and to clarify the potential smolt production of the medium and large alluvial rivers, such as Carmel, Ventura, and Santa Ynez rivers.

• Recent work shows that the tendency to outmigrate (versus mature in freshwater) is associated with particular juvenile body sizes, female sex, the presence of a particular "supergene" on chromosome Omy5, and interactions of these effects. Both variants of the supergene occur in most populations, but one variant tends to predominate in sites with connectivity to the ocean, and the other in populations without connectivity. Overall, these results show that the non-anadromous and anadromous forms are tightly integrated at the population level, suggesting a revision of the viability criterion for 100% anadromous fraction. However, such revision would require additional quantitative analysis of population viability.

• Identification of drought refugia is a pressing need according to the recovery plans, and the current drought provides a valuable opportunity to identify and characterize drought refugia. Knowledge about the distribution of drought refugia might suggest a revision of the Core 1/2/3 assignments of populations.

7 References

Abadia-Cardoso, A., D. E. Pearse, S. Jacobson, J. Marshall, D. Dalrymple, F. Kawasaki, G. Ruiz-Campos, and J. C. Garza. in press. Population genetic structure and ancestry of steelhead/rainbow trout (*Oncorhynchus mykiss*) at the extreme southern edge of their range in North America. Conservation Genetics.

Abdul-Aziz, O. I., N. J. Mantua, and K. W. Myers. 2011. Potential climate change impacts on thermal habitats of Pacific salmon (*Oncorhynchus* spp.) in the North Pacific Ocean and adjacent seas. Canadian Journal of Fisheries and Aquatic Sciences 68(9):1660−1680.

Adams, P. B., L. B. Boydstun, S. P. Gallagher, M. K. Lacy, T. McDonald, and K. E. Shaffer. 2011. California coastal salmonid population monitoring: strategy, design, and methods. California Department of Fish and Game, Fish Bulletin 180.

Allen, M. 2014. Steelhead population and habitat assessment in the Ventura River / Matilija Creek Basin 2006–2012. Normandeau Associates, Inc., Arcata, California.

Allendorf, F. W., D. Bayles, D. L. Bottom, K. P. Currens, C. A. Frissell, D. Hankin, J. A. Lichatowich, W. Nehlsen, P. C. Trotter, and T. H. Williams. 1997. Prioritizing Pacific salmon stocks for conservation. Conservation Biology 11:140−152.

Anderson, D. 2015. Unpublished data: Redwood Creek summer steelhead data 2006−2014. Compiled by D. Anderson, National Park Service, Redwood National and State Parks, Crescent City, California.

Arriaza, J. L. in review. The roles of rearing and rescue in maintaining the anadromous life history, with application to steelhead in the Carmel River. Chapter 2 of Unraveling steelhead life history complexity through mathematical modeling. Ph.D. University of California, Santa Cruz, Santa Cruz, California.

Azat, J. 2014. GrandTab 2014. California Central Valley Chinook population database report. California Department of Fish and Wildlife, Fisheries Branch. http://www.dfg.ca.gov/fish/Resources/Chinook/CValleyAssessment.asp.

Bakun, A. 1990. Global climate change and intensification of coastal ocean upwelling. Science 247:198−201.

Bakun, A., D. B. Field, A. Redondo-Rodriguez, and S. J. Weeks. 2010. Greenhouse gas, upwelling-favorable winds, and the future of coastal ocean upwelling ecosystems. Global Change Biology 16:1213−1228.

Banks, M. A., V. K. Rashbrook, M. J. Calavetta, C. A. Dean, and D. Hedgecock. 2000. Analysis of microsatellite DNA resolves genetic structure and diversity of Chinook salmon (*Oncorhynchus tshawytscha*) in California’s Central Valley. Canadian Journal of Fisheries and Aquatic Sciences 57:915−927.

Battin, J., M. W. Wiley, M. H. Ruckelshaus, R. N. Palmer, E. Korb, K. K. Bartz, and H. Imaki. 2007. Projected impacts of climate change on salmon habitat restoration. Proceedings of the National Academy of Sciences of the United States of America 104:6720−6725.

Beakes, M. P., W. H. Satterthwaite, E. M. Collins, D. R. Swank, J. E. Merz, R. G. Titus, S. M. Sogard, and M. Mangel. 2010. Smolt transformation in two California steelhead populations: effects of temporal variability in growth. Transactions of the American Fisheries Society 139:1263–1275.

Beechie, T., E. Buhle, M. Ruckelshaus, A. Fullerton, and L. Holsinger. 2006. Hydrologic regime and the conservation of salmon life history diversity. Biological Conservation 130:560–572.

Bigler, B. S., D. W. Welch, and J. H. Helle. 1996. A review of size trends among North Pacific salmon (*Oncorhynchus* spp.). Canadian Journal of Fisheries and Aquatic Sciences 53:455–465.

Bilski, R., J. Shillam, C. Hunter, M. Saldate, and E. Rible. 2013. Emigration of juvenile Chinook salmon (*Oncorhynchus tschawytscha*) and steelhead (*Oncorhynchus mykiss*) in the Lower Mokelumne River, December 2012-July 2013. East Bay Municipal Utilities District Report.

Bjorkstedt, E. P., B. C. Spence, J. C. Garza, D. G. Hankin, D. Fuller, W. E. Jones, J. J. Smith, and R. Macedo. 2005. An analysis of historical population structure for Evolutionarily Significant Units of Chinook salmon, coho salmon, and steelhead in the North-Central California Coast Recovery Domain. U.S. Department of Commerce, NOAA Technical Memorandum NMFS-SWFSC-382.

Bond, M. H. 2006. Importance of estuarine rearing to central California steelhead (*Oncorhynchus mykiss*) growth and marine survival. Master's thesis. University of California Santa Cruz, Santa Cruz, California.

Bond, M. H., S. A. Hayes, C. V. Hanson, and R. B. MacFarlane. 2008. Marine survival of steelhead (*Oncorhynchus mykiss*) enhanced by a seasonally closed estuary. Canadian Journal of Fisheries and Aquatic Sciences 65:2242–2252.

Bond, N. A., M. F. Cronin, H. Freeland, and N. J. Mantua. 2015. Causes and impacts of the 2014 warm anomaly in the NE Pacific. Geophysical Research Letter 42: 3414–3420.

Boughton, D. A. 2010a. A forward-looking scientific frame of reference for steelhead recovery on the south-central and southern California coast. U.S. Department of Commerce, NOAA Technical Memorandum NMFS-SWFSC-466.

Boughton, D. A. 2010b. Estimating the size of steelhead runs by tagging juveniles and monitoring migrants. North American Journal of Fisheries Management 30:89–101.

Boughton, D. A. 2010c. Some research questions on recovery of steelhead on the south-central and southern California coast. U.S. Department of Commerce, NOAA Technical Memorandum NMFS-SWFSC-467.

Boughton, D. A., and M. Goslin. 2006. Potential steelhead over-summering habitat in the South-Central/Southern California Coast recovery domain: Maps based on the envelope method. U.S. Department of Commerce, NOAA Technical Memorandum NMFS-SWFSC-391.

Boughton, D. A., P. Adams, E. Anderson, C. Fusaro, E. A. Keller, E. Kelley, L. Lentsch, J. Nielsen, K. Perry, H. M. Regan, J. J. Smith, C. Swift, L. Thompson, and F. Watson. 2006. Steelhead of the south-central/southern California coast: population characterization for recovery planning. U.S. Department of Commerce, NOAA Technical Memorandum NMFS-SWFSC-394.

Boughton, D. A., P. Adams, E. Anderson, C. Fusaro, E. A. Keller, E. Kelley, L. Lentsch, J. Nielsen, K. Perry, H. M. Regan, J. J. Smith, C. Swift, L. Thompson, and F. Watson. 2007. Viability criteria for steelhead of the south-central and southern California coast. U.S. Department of Commerce, NOAA Technical Memorandum NMFS-SWFSC-407.

Boughton, D. A., H. Fish, J. Pope, and G. Holt. 2009. Spatial patterning of habitat for *Oncorhynchus mykiss* in a system of intermittent and perennial streams. Ecology of Freshwater Fish 18:92−105.

Boyd, J. W., C. S. Guy, T. B. Horton, and S. A. Leathe. 2010. Effects of catch-and-release angling on salmonids at elevated water temperatures. North American Journal of Fisheries Management 30:898−907.

Buchanan, R. 2013. OCAP 2011 Steelhead tagging study: statistical methods and results. Report to J. Israel, U.S. Bureau of Reclamation, Bay Delta Office, Sacramento, California 95814.

Burke, B. J., J. J. Anderson, and A. M. Baptista. 2014. Evidence for multiple navigational sensory capabilities of Chinook salmon. Aquatic Biology 20:77−90.

Busby, P. J., T. C. Wainwright, G. J. Bryant, L. Lierheimer, R. S. Waples, F. W. Waknitz, and I. V. Lagomarsino. 1996. Status review of West Coast steelhead from Washington, Idaho, Oregon, and California. U.S. Department of Commerce, NOAA Technical Memorandum NMFS-NWFSC-27.

Busch, D. S., C. J. Harvey, and P. McElhany. 2013. Potential impacts of ocean acidification on the Puget Sound food web. ICES Journal of Marine Science 70:823−833.

Cachuma Operation and Maintenance Board. 2013. 2011 Annual monitoring summary and trend analysis for the biological opinion for the operation and maintenance of the Cachuma Project on the Santa Ynez River in Santa Barbara County, California. Cachuma Operations and Maintenance Board, Santa Barbara, California.

Caldwell, J., B. Rajagopalan, and E. Danner. 2015. Statistical modeling of daily water temperature attributes on the Sacramento River. Journal of Hydrologic Engineering 20(5): article 04014065.

CALFED Bay-Delta Program. 2000. Ecosystem restoration program plan: strategic plan for ecosystem restoration. Sacramento, California 95814.

CDFG (California Department of Fish and Game). 2002. Status review of California coho salmon north of San Francisco. Report to the California Fish and Game Commission. California Department of Fish and Game, The Resources Agency, State of California Candidate Species Status Review Report 2002-3.

California HSRG (California Hatchery Scientific Review Group). 2012. California hatchery review report. Prepared for the U.S. Fish and Wildlife Service and Pacific States Marine Fisheries Commission. June 2012.

Carlisle, S., and M. Reichmuth. 2015. Long-term monitoring of coho salmon and steelhead trout during freshwater life stages in coastal Marin County: 2013 Annual Report. Natural Resource Report NPS/SFAN/NRR—2015/956. National Park Service, Fort Collins, Colorado.

Carlson, S. M., and T. R. Seamons. 2008. A review of quantitative genetic components of fitness in salmonids: implications for adaptation to future change. Evolutionary Applications 1:222−238.

Casagrande, J. 2014. Uvas Creek juvenile steelhead distribution and abundance and adult observations, 2013. Unpublished report dated 6 February 2014. National Marine Fisheries Service, West Coast Region Office, Santa Rosa, California.

CMWD (Casitas Municipal Water District). 2013. 2013 Robles Fish Passage Facility progress report. Casitas Municipal Water District, Oak View, California.

Chesney, D., and M. Knechtle. 2015. Shasta River Chinook and coho salmon observations in 2014 Siskiyou County, CA. Final Report, Klamath River Project, 16 April 2015. California Department of Fish and Wildlife, Yreka, California.

Cheung, W. W. L., V. W. Y. Lam, J. L. Sarmiento, K. Kearney, R. Watson, and D. Pauly. 2009. Projecting global marine biodiversity impacts under climate change scenarios. Fish and Fisheries 10:235−251.

Cheung, W. W. L., R. D. Brodeur, T. A. Okey, and D. Pauly. 2015. Projecting future changes in distributions of pelagic fish species of Northeast Pacific shelf seas. Progress in Oceanography 130:19−31.

Clemento, A. J., E. C. Anderson, D. Boughton, D. Girman, and J. C. Garza. 2009. Population genetic structure and ancestry of *Oncorhynchus mykiss* populations above and below dams in south-central California. Conservation Genetics 10:1321−1336.

Clifford, M. 2015. Unpublished data: Warm Springs Hatchery production history. California Department of Fish and Wildlife, Mt. Shasta, California.

Coey, B. 2015. Unpublished data: spreadsheet, Russian River adult returns, 1981−2015. National Marine Fisheries Service, West Coast Region Office, Santa Rosa, California.

Confer, T. 2015. Unpublished data: spreadsheet, Huntley Park summary 1974 – (revised May 2015). Oregon Department of Fish and Wildlife, Gold Beach, Oregon.

Cook, E. R., C. A. Woodhouse, C. M. Eakin, D. M. Meko, and D. W. Stahle. 2004. Long-term aridity changes in the western United States. Science 306:1015−1018.

Crozier, L. G., A. P. Hendry, P. W. Lawson, T. P. Quinn, N. J. Mantua, J. Battin, R. G. Shaw, and R. B. Huey. 2008. Potential responses to climate change in organisms with complex life histories: evolution and plasticity in Pacific salmon. Evolutionary Applications 1:252−270.

Crozier, L. G., and J. A. Hutchings. 2014. Plastic and evolutionary responses to climate change in fish. Evolutionary Applications 7:68−87.

Crozier, L. G., and R. W. Zabel. 2006. Climate impacts at multiple scales: evidence for differential population responses in juvenile Chinook salmon. Journal of Animal Ecology 75:1100−1109.

Crozier, L., R. W. Zabel, S. Achord, and E. E. Hockersmith. 2010. Interacting effects of density and temperature on body size in multiple populations of Chinook salmon. Journal of Animal Ecology 79:342−349.

Cuthbert, R., P. Cuthbert, A. Fuller, and M. Hellmair. 2014a. Salinas River basin adult steelhead escapement monitoring, 2014 Annual Report. FISHBIO, Oakdale, California.

Cuthbert, R., P. Cuthbert, A. Fuller, and M. Hellmair. 2014b. Salinas basin juvenile *O. mykiss* downstream migration monitoring. FISHBIO, prepared for Monterey County Water Resources Agency, Oakdale, California.

Dagit, R., K. Adamek, S. Albers, E. Montgomery, and A. Sanchez. 2015. Topanga Creek steelhead monitoring, March 2011 − December 2014. Prepared for CDFW Contract No. P1050009. Resource Conservation District of the Santa Monica Mountains, Topanga, California.

DeHaven, R.W. 2010. Adult and juvenile steelhead population surveys, Gualala River, California, 2009. (available online at: http://vme.net/dvm/gualala-river/pdf/2010GRSSAnRepOnly.pdfvme.net/dvm/gualala-river/pdf/2009GRSSAnRepOnly-1.pdf).

Diffenbaugh, N. S., F. Giorgi, and J. S. Pal. 2008. Climate change hotspots in the United States. Geophysical Research Letters 35:L16709.

Doney, S. C., L. Bopp, and M. C. Long. 2014. Historical and future trends in ocean climate and biogeochemistry. Oceanography 27:108−119.

Donohoe, C. J., P. B. Adams, and C. F. Royer. 2008. Influence of water chemistry and migratory distance on ability to distinguish progeny of sympatric resident and anadromous rainbow trout (*Oncorhynchus mykiss*). Canadian Journal of Fisheries and Aquatic Sciences 65:1060−1075.

Donohoe, C. J., and R. Null. 2013. Migratory history and maternal origin of rainbow trout (*Oncorhynchus mykiss*) returning to Coleman National Fish Hatchery in 2008. Report to U.S. Fish and Wildlife Service, Red Bluff Fish and Wildlife Office, California.

Duffy, W. 2013. Unpublished data: USGS California Cooperative Fisheries Research Unit, Humboldt State University, Arcata, California.

EBMUD (East Bay Municipal Utility District). 2011. Lower Mokelumne River upstream fish migration monitoring conducted at Woodbridge Irrigation District Dam August 2010−July 2011. East Bay Municipal Utility District, Lodi, California.

EBMUD (East Bay Municipal Utility District). 2012. Lower Mokelumne River upstream fish migration monitoring conducted at Woodbridge Irrigation District Dam August 2011−July 2012. East Bay Municipal Utility District, Lodi, California.

EBMUD (East Bay Municipal Utility District). 2013. Lower Mokelumne River upstream fish migration monitoring conducted at Woodbridge Irrigation District Dam August 2012−July 2013. East Bay Municipal Utility District, Lodi, California.

Ettlinger, E., G. Andrew, P. Doughty, and V. Rogers. 2015. Adult salmonid monitoring in the Lagunitas Creek Watershed – 2014−2015. Report to California Department of Fish and Wildlife, Fisheries Restoration Grant Program, Grant # P1130417. Marin Municipal Water District, Corte Madera, California. (available online at: https://www.marinwater.org/DocumentCenter/View/3429http://www.marinwater.org/documents/MMWD_Spawner_Report_2008_2009.pdf).

Farhat, B. in preparation. State-wide estimates of steelhead angler effort, 2007−2014. California Department of Fish and Wildlife, Sacramento, California.

Fisher, F. W. 1994. Past and present status of Central Valley Chinook salmon. Conservation Biology 8:870−873.

Fisher, J. P., L. A. Weitkamp, D. J. Teel, S. A. Hinton, J. A. Orsi, E. V. Farley Jr., J. F. T. Morris, M. E. Thiess, R. M. Sweeting, and M. Trude. 2014. Early ocean dispersal patterns of Columbia River Chinook and coho salmon. Transactions of the American Fisheries Society 143:252−272.

Ford, M. (editor). 2011. Status review update for Pacific salmon and steelhead listed under the Endangered Species Act: Northwest. U.S. Department of Commerce, NOAA Technical Memorandum NOAA-NWFSC-113.

Ford, T., and S. Kirihara. 2010. Tuolumne River *Oncorhynchus mykiss* monitoring report. Prepared by Turlock Irrigation District/Modesto Irrigation District, California and Stillwater Sciences, Berkeley, California for Federal Energy Regulatory Commission, Washington, D.C.

Franks, S. 2012. Possibility of natural producing spring-run Chinook salmon in the Stanislaus and Tuolumne rivers. Memo, Central Valley Area Office, November 2012.

Fry, D. H., Jr. 1961. King salmon spawning stocks of the California Central Valley, 1940-1959. California Fish and Game 47:55−71.

Fuller, A. N. 2008. Outmigrant trapping of juvenile salmonids in the lower Tuolumne River, 2007. Prepared by FISHBIO Environmental, Chico, California for Turlock and Modesto Irrigation Districts.

Fuller, A. N. 2015. Unpublished data: table of the estimated number of adult steelhead observed in the video weir on the Stanislaus River 2003−2014. FISHBIO, Oakdale, California.

Gallagher, S. P., and C. M. Gallagher. 2005. Discrimination of Chinook salmon, coho salmon, and steelhead redds and evaluation of the use of redd data for estimating escapement in several unregulated streams in northern California. North American Journal of Fisheries Management 25:284−300.

Gallagher, S. P., Thompson, S., and D. W. Wright. 2014. Coastal Mendocino County salmonid life cycle and regional monitoring: monitoring status and trends for 2013. 2012−13 Administrative Report. California State Department of Fish and Wildlife, Coastal Watershed Planning and Assessment Program, 1487 Sandy Prairie Court, Suite A, Fortuna, California.

Garfin, G., G. Franco, H. Blanco, A. Comrie, P. Gonzalez, T. Piechota, R. Smyth, and R. Waskom. 2014. Southwest. Pages 462-486 *in* J. M. Melillo, T. C. Richmond, and G. W. Yohe, editors. Climate change impacts in the United States: the Third National Climate Assessment. U.S. Global Change Research Program. http://nca2014.globalchange.gov/system/files_force/downloads/low/NCA3_Full_Report_20_Southwest_LowRes.pdf?download=1

Garmin, C. E. 2015. Butte Creek Chinook salmon, *Oncorhynchus tshawytscha* pre-spawn mortality evaluation 2014. California Department of Fish and Wildlife. Inland Fisheries, Report Number 2015-1.

Garwood, J. 2012. Historic and recent occurrence of coho salmon (*Oncorhynchus kisutch*) in California streams within the Southern Oregon/Northern California Evolutionarily Significant Unit. California Department of Fish and Game, Fisheries Branch Administrative Report, 2012. Northern Region Anadromous Fisheries Resource and Monitoring Program, 5341 Ericson Way, Arcata, California.

Garwood, J. M., and M. D. Larson. 2014. Reconnaissance of salmonid redd abundance and juvenile salmonid spatial structure in the Smith River with emphasis on coho salmon (*Oncorhynchus kisutch*). Final Report to the California Department of Fish and Wildlife, Fisheries Restoration Grants Program, Grantee Agreement P1010504.

Garza, J. C., and D. E. Pearse. 2008. Population genetics of *Oncorhynchus mykiss* in the Santa Clara Valley Region. Report to the Santa Clara Valley Water District. National Marine Fisheries Service, Southwest Fisheries Science Center, 110 Shaffer Road, Santa Cruz, California.

Garza, J. C., L. Gilbert-Horvath, B. Spence, T. H. Williams, J. Anderson, and H. Fish. 2014. Population structure of steelhead in coastal California. Transactions of the American Fisheries Society 143:134−152.

Gilbert-Horvath, E. A., K. Pipal, B. Spence, T. Williams, and J. C. Garza. in press. Hierarchical phylogeographic structure of Coho Salmon (*Oncorhynchus kisutch*) in California. Transactions of the American Fisheries Society.

Goin, M. 2014. A 2013–2014 escapement estimate for Central California Coast Coho Salmon (*Oncorhynchus kisutch*) and Steelhead (*Oncorhynchus mykiss*) in coastal San Mateo and Santa Cruz counties. Report submitted to the California Department of Fish and Wildlife, as part of the requirements of FRGP Grant Agreement Number P1230418 00.

Goin, M. 2015 (draft). 2014–2015 escapement estimates for Central California Coast Coho Salmon (*Oncorhynchus kisutch*) and Steelhead (*Oncorhynchus mykiss*) in coastal San Mateo and Santa Cruz counties. Report submitted to the California Department of Fish and Wildlife, as part of the requirements of FRGP Grant Agreement Number P1230418 00.

Good, T. P., R. S. Waples, and P. Adams (editors). 2005. Updated status of federally listed ESUs of West Coast salmon and steelhead. U.S. Department of Commerce, NOAA Technical Memorandum NMFS-NWFSC-66.

Grover, A., A. Low, P. Ward, J. Smith, M. Mohr, D. Viele, and C. Tracy. 2004. Recommendations for developing fishery management plan conservation objectives for Sacramento River winter Chinook and Sacramento River spring Chinook. Progress Report, March 2014.

Gruber, N., C. Hauri, Z. Lachkar, D. Loher, T. L. Frölicher, and G. Plattner. 2012. Rapid progression of ocean acidification in the California Current System. Science 337:220‒223.

Guignard, J. 2015. Unpublished data: Spreadsheet, number of adult Chinook salmon observed in the video weir between February‒June on the Stanislaus River. FISHBIO, Oakdale, California.

Harris, S. 2010. Annual project performance report. Project 56, Job 3. California Department of Fish and Game, Willits, California.

Harris, S., and S. Thompson. 2013. Annual project performance report. Anadromous Sport Fish Management and Research, F-137-R-2. Project 56 and 56a: North Central District salmon and steelhead management. California Department of Fish and Wildlife.

Harris, S., and S. Thompson. 2014. Final project performance report. Northern Region Anadromous Sport Fish Management and Research, Federal Award No. F13AF00827 (F137, P.56). Project 56 and 56a: North Central District salmon and steelhead management (G1398060).

Hassrick, J. L., M. J. Henderson, D. D. Huff,, W. J. Sydeman, M. C. Sabal, J. A. Harding, A. J. Ammann, E. D. Crandall, E. Bjorkstedt, J. C. Garza, and S. A. Hayes. 2016. Early ocean distribution of juvenile Chinook salmon in an upwelling ecosystem. Fisheries Oceanography 25:133‒146.

HSRG (Hatchery Scientific Review Group). 2014. On the science of hatcheries: an updated perspective on the role of hatcheries in salmon and steelhead management in the Pacific Northwest. A. Appleby, H. L. Blankenship, D. Campton, K. Currens, T. Evelyn, D. Fast, T. Flagg, J. Gislason, P. Kline, C. Mahnken, B. Missildine, L. Mobrand, G. Nandor, P. Paquet, S. Patterson, L. Seeb, S. Smith, and K. Warheit. June 2014.

Hauri, C., N. Gruber, G. K. Plattner, S. Alin, R. A. Feely, B. Hales, and P. A. Wheeler. 2009. Ocean Acidification in the California Current System. Oceanography 22:60‒71.

Hayes, D. B., B. J. Bellgraph, B. M. Roth, D. D. Dauble, and R. P. Mueller. 2014. Timing of redd construction by fall Chinook salmon in the Hanford Reach of the Columbia River. River Research and Applications 30:1110‒1119.

Healey, M. 2011. The cumulative impacts of climate change on Fraser River sockeye salmon (*Oncorhynchus nerka*) and implications for management. Canadian Journal of Fisheries and Aquatic Sciences 68:718‒737.

Hedgecock, D. E., M. A. Banks, V. K. Rashbrook, C. A. Dean, and S. M. Blankenship. 2001. Applications of population genetics to conservation of Chinook salmon diversity in the Central Valley. Pages 45−70 *in* R. L. Brown, editor. Contributions to the biology of Central Valley salmonids. California Department of Fish and Wildlife, Fish Bulletin 179.

Hendrix, N., A. Criss, E. Danner, C. M. Green, C. M., H. Imaki, A. Pike, and S. T. Lindley. 2014. Life cycle modeling framework for Sacramento River Winter-run Chinook salmon. U.S. Department of Commerce, NOAA Technical Memorandum NMFS-SWFSC-530.

Holloway, W., S. P. Gallagher, S. Thompson, D. W. Wright, and E. Lang. 2014. Coastal Mendocino County salmonid life cycle and regional monitoring: monitoring status and trends for 2014. 2013−2014 Administrative Report. Pacific States Marine Fisheries Commission and California State Department of Fish and Wildlife, Coastal Watershed Planning and Assessment Program, Fortuna, California.

Horton, G. 2015. Unpublished data: Russian River Chinook salmon 2000-2014. Sonoma County Water Agency, Santa Rosa, California.

Howard, S., and M. Booth. 2013. Fish passage monitoring and studies, Freeman Diversion Facility, Santa Clara River, Ventura County, California. Annual Report, 2013 Monitoring Season. United Water Conservation District, Santa Paula, California.

IPCC (Intergovernmental Panel on Climate Change). 2013. Climate change 2013: the physical science basis. Contribution of Working Group I to the Fifth Assessment report of the Intergovernmental Panel on Climate Change. Cambridge University Press. Available from: http://www.climatechange2013.org/ Cambridge, United Kingdom and New York, NY, USA.

IPCC (Intergovernmental Panel on Climate Change). 2014. Climate change 2014: synthesis report. Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. IPCC, Geneva, Switzerland.

ISAB (Independent Scientific Advisory Board). 2007. Climate change impacts on Columbia River basin fish and wildlife. Northwest Power and Conservation Council, Columbia River Basin Indian Tribes, National Marine Fisheries Service, Portland, Oregon.

Jackson, T. A. 2007. California steelhead fishing report-restoration card: a report to the legislature. California Department of Fish and Game, Sacramento, California.

Jacobson, S., J. Marshall, D. Dalrymple, F. Kawasaki, D. E. Pearse, A. Abadia-Cardoso, and J. C. Garza. 2014. Genetic analysis of trout (*Oncorhynchus mykiss*) in southern California coastal rivers and streams. Final Report for California Department of Fish and Wildlife, Fisheries Restoration Grant Program, Project No. 0950015.

Jacox, M. G., A. M. Moore, C. A. Edwards, and J. Fiechter. 2014. Spatially resolved upwelling in the California Current System and its connections to climate variability. Geophysical Research Letters 41:3189−3196.

Jankovitz, J. 2013. 2012–2013 escapement estimates for Central California Coast Coho Salmon (*Oncorhynchus kisutch*) and Steelhead (*Oncorhynchus mykiss*) south of the Golden Gate. Prepared by Pacific States Marine Fisheries Commission in cooperation with California Department of Fish and Wildlife, Region 3.

Johnstone, J. A., and N. J. Mantua. 2014. Atmospheric controls on northeast Pacific temperature variability and change, 1900−2012. Proceedings of the National Academy of Sciences 111:14360-14365.

Johnstone, J. A., and T. E. Dawson. 2010. Climatic context and ecological implications of summer fog decline in the coast redwood region. Proceedings of the National Academy of Sciences 107:4533−4538.

Kendall, N. W., J. R. McMillan, M. R. Sloat, T. W. Buehren, T. P. Quinn, G. R. Pess, K. V. Kuzishchin, M. M. McClure, and R. W. Zabel. 2015. Anadromy and residency in steelhead and rainbow trout (*Oncorhynchus mykiss*): a review of the processes and patterns. Canadian Journal of Fisheries and Aquatic Sciences 72:319−342.

Kiernan, J. 2015. Unpublished data: escapement data for steelhead and coho salmon from Scott Creek, Santa Cruz County, California. National Marine Fisheries Service, Southwest Fisheries Science Center, Santa Cruz, California.

Killiam, D. 2014. Unpublished data: information summary table on winter-run Chinook salmon carcass survey data and some RBDD (Red Bluff Diversion Dam) for 1996 to present. California Department of Fish and Wildlife, Redding, California.

Killam D., M. Johnson, and R. Revnak. 2014. Chinook salmon populations of the upper Sacramento River Basin in 2013. California Department of Fish and Wildlife, RBFO Technical Report No. 02-2014.

Klos, P. Z., T. E. Link, and J. T. Abatzoglou. 2014. Extent of the rain-snow transition zone in the western U.S. under historic and projected climate. Geophysical Research Letters 41:4560−4568.

Knechtle, M., and D. Chesney. 2014. 2013 Scott River salmon studies, final report. California Department of Fish and Wildlife, Yreka, California.

Knechtle, M. 2015. Unpublished data: spreadsheet, video weir time series data for coho salmon, Scott and Shasta rivers. California Department of Fish and Wildlife, Klamath River Project, Yreka, California.

Koehler, J. T. 2008. Napa Creek salmon monitoring project. Annual Report – Year 3 (June 2007–June 2008). Prepared for the City of Napa, Agreement #8981. Napa County Resource Conservation District, Napa, California.

Koehler, J. T. 2014. Napa River steelhead and salmon monitoring program: 2014 rotary screw trap results. Technical Memorandum, Napa County Resource Conservation District, Napa, California.

Koehler, J. T., and P. D. Blank. 2013. Napa River steelhead and salmon monitoring report: 2012−2013 season. Napa County Resource Conservation District, Napa, California.

Kormos, B., M. Palmer-Zwhalen, and A. Low. 2012. Recovery of coded-wire tags from Chinook salmon in California's Central Valley escapement and ocean harvest in 2010. Fisheries Branch Administrative Report 2012-02.

Lawrence, D. J., B. Stewart-Koster, J. D. Olden, A. S. Ruesch, C. E. Torgersen, J. J. Lawler, D. P. Butcher, and J. K. Crown. 2014. The interactive effects of climate change, riparian management, and a nonnative predator on stream-rearing salmon. Ecological Applications 24:895−912.

Lawrence, D. J., J. D. Olden, and C. E. Torgersen. 2012. Spatiotemporal patterns and habitat associations of smallmouth bass (*Micropterus dolomieu*) invading salmon-rearing habitat. Freshwater Biology 57:1929−1946.

Leduc, A., P. L. Munday, G. E. Brown, and M. C. O. Ferrari. 2013. Effects of acidification on olfactory-mediated behaviour in freshwater and marine ecosystems: A synthesis. Philosophical Transactions of the Royal Society B-Biological Sciences 368:20120447.

Levin, P. S. 2003. Regional differences in responses of Pacific Salmon to large-scale climate patterns. Journal of Biogeography 30:711−717.

Lindley, S. T., R. Schick, B. P. May, J. J. Anderson, S. Greene, C. Hanson, A. Low, D. McEwan, R. B. MacFarlane, C. Swanson, and J. G. Williams. 2004. Population structure of threatened and endangered Chinook salmon ESUs in California's Central Valley basin. U.S. Department of Commerce, NOAA Technical Memorandum NMFS-SWFSC-360.

Lindley, S. T., R. S. Schick, A. Agrawal, M. Goslin, T. E. Pearson, E. Mora, J. Anderson, B. May, S. Greene, C. Hanson, A. Low, D. McEwan, R. B. MacFarlane, C. Swanson, and J. G. Williams. 2006. Historical population structure of Central Valley steelhead and its alteration by dams. San Francisco Estuary and Watershed Science 4(1): article 3.

Lindley, S. T., R. S. Schick, E. Mora, P. B. Adams, J. J. Anderson, S. Greene, C. Hanson, B. May, D. McEwan, R. B. MacFarlane, C. Swanson, and J. G. Williams. 2007. Framework for assessing viability of threatened and endangered Chinook salmon and steelhead in the Sacramento-San Joaquin Basin. San Francisco Estuary and Watershed Science 5(1): article 4.

Litzow, M. A., F. J. Mueter, and A. J. Hobday. 2014. Reassessing regime shifts in the North Pacific: incremental climate change and commercial fishing are necessary for explaining decadal-scale biological variability. Global Change Biology 20:38−50.

Marin County Watershed Program. 2015. Miller Creek Watershed. (Available online: http://www.marinwatersheds.org/miller_creek.html

Martinez, A., J. C. Garza, and D. E. Pearse. 2011. A microsatellite genome screen identifies chromosomal regions under differential selection in steelhead and rainbow trout. Transactions of the American Fisheries Society 140:829−842.

Materna, E. 2001. Temperature Interaction, Issue Paper 4. Environmental Protection Agency Region 10, U.S. Fish and Wildlife Service, EPA-910-D-01-004, Seattle, Washington.

MSG (Mattole Salmon Group). 2015. Summer steelhead dive counts. Mattole Salmon Group, Petrolia, California. (data derived from graphs available online at http://www.mattolesalmon.org/programs/fisheries/monitoring/summer-steelhead-dive/)

McClure, M., M. Alexander, D. Borggaard, D. Boughton, L. Crozier, R. Griffis, J. C. Jorgensen, S. T. Lindley, J. Nye, M. J. Rowland, E. E. Seney, A. Snover, C. Toole, and K. Van Houtan. 2013. Incorporating climate science in applications of the U.S. Endangered Species Act for aquatic species. Conservation Biology 27:1222−1233.

McElhany, P., M. H. Ruckelshaus, M. J. Ford, T. C. Wainwright, and E. P. Bjorkstedt. 2000. Viable salmonid populations and the recovery of evolutionarily significant units. U.S. Department of Commerce, NOAA Technical Memorandum NMFS-NWFSC-42.

Melillo, J. M., T. C. Richmond, and G. W. Yohe, editors. 2014. Climate change impacts in the United States: The Third National Climate Assessment. U.S. Global Change Research Program. http://nca2014.globalchange.gov/downloads

Miller, K. M., A. Teffer, S. Tucker, S. Li, A. D. Schulze, M. Trudel, F. Juanes, A. Tabata, K. H. Kaukinen, N. G. Ginther, T. J. Ming, S. J. Cooke, J. M. Hipfner, D. A. Patterson, and S. G. Hinch. 2014. Infectious disease, shifting climates, and opportunistic predators: cumulative factors potentially impacting wild salmon declines. Evolutionary Applications 7:812−855.

Monterey County Water Resources Agency. 2014. Arroyo Seco and Nacimiento rivers index reach monitoring: dive count surveys. Monterey County Water Resources Agency, Salinas, California.

Morita, K., S. H. Morita, M. Fukuwaka, and H. Matsuda. 2005. Rule of age and size at maturity of chum salmon (*Oncorhynchus keta*): implications of recent trends among *Oncorhynchus* spp. Canadian Journal of Fisheries and Aquatic Sciences 62:2752−2759.

Morrison, A., K. Bogier, and B. Keimel. 2014. Petaluma Watershed steelhead monitoring report—2013–2014 spawning surveys. National Marine Fisheries Service, Santa Rosa, California.

Mote, P. W., and N. J. Mantua. 2002. Coastal upwelling in a warmer future. Geophysical Research Letters 29:2138.

Mote, P., A. K. Snover, S. Capalbo, S. D. Eigenbrode, P. Glick, J. Littell, R. Raymondi, and S. Reeder. 2014. Northwest. Pages 487−513 *in* J. M. Melillo, T. C. Richmond, and G. W. Yohe, editors. Climate change impacts in the United States: The Third National Climate Assessment. U.S. Global Change Research Program. http://nca2014.globalchange.gov/system/files_force/downloads/low/NCA3_Full_Report_21_Northwest_LowRes.pdf?download=1

Moyle, P. B., J. D. Kiernan, P. K. Crain, and R. M. Quinones. 2013. Climate change vulnerability of native and alien freshwater fishes of California: a systematic assessment approach. PLoS ONE 8(5).

MPWMD (Monterey Peninsula Water Management District). 2013. 2011-2012 Annual Report, Monterey Peninsula Water Management District, Mitigation Program, Water Allocation Program EIR. Monterey, California.

Mueter, F. J., R. M. Peterman, and B. J. Pyper. 2002. Opposite effects of ocean temperature on survival rates of 120 stocks of Pacific salmon (*Oncorhynchus* spp.) in northern and southern areas. Canadian Journal of Fisheries and Aquatic Sciences 59:456−463.

Mueter, F. J., B. J. Pyper, and R. M. Peterman. 2005. Relationships between coastal ocean conditions and survival rates of northeast Pacific salmon at multiple lags. Transactions of the American Fisheries Society 134:105−119.

Myers, J. M., R. G. Kope, G. J. Bryant, D. Teel, L. J. Lierheimer, T. C. Wainwright, S. W. Grant, W. F. Waknitz, K. Neely, S. T. Lindley, and R. S. Waples. 1998. Status review of Chinook salmon from Washington, Idaho, Oregon, and California. U.S. Department of Commerce, National Marine Fisheries Service, NOAA Technical Memorandum NMFS-NWFSC-35.

Naik, P. K., and D. A. Jay. 2011a. Distinguishing human and climate influences on the Columbia River: Changes in mean flow and sediment transport. Journal of Hydrology 404:259−277.

Naik, P. K., and D. A. Jay. 2011b. Human and climate impacts on Columbia River hydrology and salmonids. River Research and Applications 27:1270−1276.

NMFS (National Marine Fisheries Service). 1997. Status update for West Coast steelhead from Washington, Idaho, Oregon, and California. Memorandum dated 7 July 1997 from the Biological Review Team to the National Marine Fisheries Service Northwest Regional Office, Portland, Oregon.

NMFS (National Marine Fisheries Service). 1999. Status review update for deferred ESUs of West Coast Chinook salmon (*Oncorhynchus tshawytscha*) from Washington, Oregon, California, and Idaho. Memorandum dated 16 July 1999 to U. Varanasi, Northwest Fisheries Science Center, and M. Tillman, Southwest Fisheries Science Center, from M. Schiewe, Northwest Fisheries Science Center, Montlake, Washington.

NMFS (National Marine Fisheries Service). 2002. Biological opinion for the proposed license for amendment for the Potter Valley Project (Federal Energy Regulatory Commission Project Number 77-110). Endangered Species Act Section 7 Consultation prepared by NMFS Southwest Region, Santa Rosa, California.

NMFS (National Marine Fisheries Service). 2009a. Biological and conference opinion on the long-term operations of the Central Valley Project and State Water Project. Endangered Species Act Section 7 Consultation prepared by NMFS Central Valley Office, Sacramento, California.

NMFS (National Marine Fisheries Service). 2009b. Southern California Steelhead Recovery Plan - Public Review Draft Version July 2009. Southwest Regional Office, NMFS, Long Beach, California. (www.nmfs.gov/pr/pdfs/recovery/draft_steelhead_socal.pdf)

NMFS (National Marine Fisheries Service). 2010. Endangered Species Act Section 7 Consultation Biological Opinion: Authorization of Ocean Salmon Fisheries Pursuant to the Pacific Coast Salmon Fishery Management Plan and Additional Protective Measures as it affects Sacramento River Winter Chinook Salmon. National Marine Fisheries Service, Southwest Region, Protected Resources Division. April 30, 2010.

NMFS (National Marine Fisheries Service). 2012a. Final Recovery Plan for Central California Coast coho salmon Evolutionarily Significant Unit. National Marine Fisheries Service, Southwest Region, Santa Rosa, California.

NMFS (National Marine Fisheries Service). 2012b. Southern California Steelhead Recovery Plan. Southwest Region, Protected Resources Division, Long Beach, California.

NMFS (National Marine Fisheries Service). 2014a. Final Recovery Plan for the Southern Oregon/Northern California Coast Evolutionarily Significant Unit of Coho Salmon (*Oncorhynchus kisutch*). National Marine Fisheries Service, Arcata, California.

NMFS (National Marine Fisheries Service). 2014b. Recovery Plan for the Evolutionarily Significant Units of Sacramento River Winter-run Chinook Salmon and Central Valley Spring-run Chinook Salmon and the Distinct Population Segment of California Central Valley Steelhead. National Marine Fisheries Service, California Central Valley Area Office, Sacramento, California.

NMFS (National Marine Fisheries Service). 2015. November takes a bite out of the blob. https://swfsc.noaa.gov/news.aspx?ParentMenuId=54&id=21508

National Research Council. 2012. Sea-level rise for the coasts of California, Oregon, and Washington: past, present, and future. The National Academies Press, USA. (Available at: http://www.nationalacademies.org/nrc/)

Northwest Fisheries Science Center. 2015. Status review update for Pacific salmon and steelhead listed under the Endangered Species Act: Pacific Northwest. National Marine Fisheries Service, Northwest Fisheries Science Center, Seattle, Washington.

O’Farrell, M. R., and W. H. Satterthwaite. 2015. Inferred historical fishing mortality rates for an endangered population of Chinook salmon (*Oncorhynchus tshawytscha*). Fishery Bulletin 113:341−351.

O’Farrell, M. R., W. H. Satterthwaite, and B. C. Spence. 2012. California Coastal Chinook salmon: status, data, and feasibility of alternative fishery management strategies. NOAA Technical Memorandum NMFS-SWFSC-494.

Ohms, H. A., M. R. Sloat, G. H. Reeves, C. E. Jordan, and J. B. Dunham. 2014. Influence of sex, migration distance, and latitude on life history expression in steelhead and rainbow trout (*Oncorhynchus mykiss*). Canadian Journal of Fisheries and Aquatic Sciences 71:70−80.

ODFW (Oregon Department of Fish and Wildlife). 2015. Cole Rivers Hatchery, operations plan 2015. http://www.dfw.state.or.us/fish/HOP/Cole%20Rivers%20HOP.pdf

Otero, J., J. H. L'Abée-Lund, T. Castro-Santos, K. Leonardsson, G. Storvik, B. Jonsson, B. Dempson, I. C. Russell, A. J. Jensen, J. Bagliniére, M. Dionne, J. D. Armstrong, A. Romakkaniemi, B. H. Letcher, J. F. Kocik, J. Erkinaro, R. Poole, G. Rogan, H. Lundqvist, J. C. MacLean, E. Jokikokko, J. Arnekleiv, R. J. Kennedy, E. Niemela, P. Caballero, P. A. Music, T. Antonsson, S. Gudjonsson, A. E. Veselov, A. Lamberg, S. Groom, B. H. Taylor, M. Taberner, M. Dillane, F. Arnason, G. Horton, N. A. Hvidsten, I. R. Jonsson, N. Jonsson, S. McKelvey, T. F. Næsje, Ø. Skaala, G. W. Smith, H. Sægrov, N. C. Stenseth, and L. A. Vøllestad. 2014. Basin-scale phenology and effects of climate variability on global timing of initial seaward migration of Atlantic salmon (*Salmo salar*). Global Change Biology 20:61−75.

Ou, M., T. J. Hamilton, J. Eom, E. M. Lyall, J. Gallup, A. Jiang, J. Lee, D. A. Close, S. Yun, and C. J. Brauner. 2015. Responses of pink salmon to CO_2-induced aquatic acidification. Nature Climate Change 5:950−955.

PFMC (Pacific Fishery Management Council). 2015a. Review of 2014 ocean salmon fisheries: stock assessment and fishery evaluation document for the Pacific Coast Salmon Fishery Management Plan. Appendix B, Table B-7.

PFMC (Pacific Fishery Management Council). 2015b. Preseason report I: stock abundance analysis and environmental assessment Part 1 for 2015 ocean salmon fishery regulations. (Document prepared for the Council and its advisory entities.) Pacific Fishery Management Council, 7700 NE Ambassador Place, Suite 101, Portland, Oregon.

PFMC (Pacific Fishery Management Council). 2016. Letter from C. Tracy to T Howard and R. Milligan, 22 April 2016; RE: Conservation of Sacramento River Chinook Salmon.

Palmer-Zwhalen, M. and B. Kormos. 2013. Recovery of coded-wire tags from Chinook salmon in California's Central Valley escapement and ocean harvest in 2011. Fisheries Branch Administrative Report 2013-02.

Pearcy, W. G. 2002. Marine nekton off Oregon and the 1997−98 El Niño. Progress in Oceanography 54:399−403.

Pearse, D. and J. C. Garza. 2015. You can't unscramble an egg: population genetic structure of *Oncorhynchus mykiss* in the California Central Valley inferred from combined microstatellite and single nucleotide polymorphism data. San Francisco Estuary and Watershed Science 13(4).

Pearse, D. E., E. C. Anderson, J. C. Garza, D. E. Rundio, T. H. Williams, and S. T. Lindley. in preparation. A simple genetic basis for individual migratory tendency in rainbow trout. National Marine Fisheries Service, Southwest Fisheries Science Center, Santa Cruz, California.

Pearse, D. E., M. R. Miller, A. Abadia-Cardoso, and J. C. Garza. 2014. Rapid parallel evolution of standing variation in a single, complex, genomic region is associated with life history in steelhead/rainbow trout. Proceedings of the Royal Society B-Biological Sciences [online serial] 281:article 20140012.

Petersen, J. H., and J. F. Kitchell. 2001. Climate regimes and water temperature changes in the Columbia River: bioenergetic implications for predators of juvenile salmon. Canadian Journal of Fisheries and Aquatic Sciences 58:1831−1841.

Peterson, J. O., C. A. Morgan, W. T. Peterson, and E. Di Lorenzo. 2013. Seasonal and interannual variation in the extent of hypoxia in the northern California Current from 1998−2012. Limnology and Oceanography 58:2279−2292.

Petrosky, C. E., and H. A. Schaller. 2010. Influence of river conditions during seaward migration and ocean conditions on survival rates of Snake River Chinook salmon and steelhead. Ecology of Freshwater Fish 19:520−536.

Pipal, K., M. Jessop, D. Boughton, and P. Adams. 2010a. Using dual-frequency identification sonar (DIDSON) to estimate adult steelhead escapement in the San Lorenzo River, California. California Fish and Game 96:90−95.

Pipal, K., M. Jessop, G. Holt, and P. Adams. 2010b. Operation of dual-frequency identification sonar (DIDSON) to monitor adult steelhead (*Oncorhynchus mykiss*) in the central California coast. NOAA Technical Memorandum NMFS-SWFSC-454.

Pipal, K. A., J. J. Notch, S. A. Hayes, and P. B. Adams. 2012. Estimating escapement for a low-abundance steelhead population using dual-frequency identification sonar (DIDSON). North American Journal of Fisheries Management 32:880−893.

Potter, C. 2014. Understanding climate change on the California coast: Accounting for extreme daily events among long-term trends. Climate 2:18−27.

Pounds, J., P. Righter, S. Gough, M. Gilroy, S. Naman, and D. Feral. 2015. Mad River summer steelhead report: 2014. (Available from Dave Feral, Mad River Alliance, at madriveralliance@suddenlink.net)

Poytress, W. R., J. Gruber, F. D. Carrillo and S. D. Voss. 2014. Compendium report of Red Bluff Diversion Dam rotary trap juvenile anadromous fish production indices for years 2002−2012. U.S. Fish and Wildlife Service to California Department of Fish and Wildlife and U.S. Bureau of Reclamation.

Poytress, W. R. 2016. Brood-year 2014 winter Chinook juvenile production indices with comparisons to juvenile production estimates derived from adult escapement. U.S. Fish and Wildlife Service, Red Bluff Fish and Wildlife Office, Red Bluff, California.

Pyper, B. J., and R. M. Peterman. 1999. Relationship among adult body length, abundance, and ocean temperature for British Columbia and Alaska sockeye salmon (*Oncorhynchus nerka*), 1967−1997. Canadian Journal of Fisheries and Aquatic Sciences 56:1716−1720.

Quinn, T. P. 2005. The behavior and ecology of Pacific salmon and trout. University of Washington Press, Seattle, Washington.

Quiñones, R. M., and P. B. Moyle. 2014. Climate change vulnerability of freshwater fishes of the San Francisco Bay area. San Francisco Estuary and Watershed Science 12(3).

Reichmuth, M. 2015. Unpublished data: coho salmon spawner surveys in Olema, Cheda, Redwood, and Pine Gulch creeks (Marin County), 1997−2014. National Park Service, Point Reyes National Seashore. Point Reyes Station, California.

Ricker, S. J. 2015. Unpublished data: Freshwater Creek 2014−2015 Life cycle monitoring station: summary results for National Marine Fisheries Service, coho salmon 5-year status update. California Department of Fish and Wildlife, Anadromous Fisheries Resource Assessment and Monitoring Program, Arcata, California.

Ricker, S. J., and C. W. Anderson. 2014. Results of Freshwater Creek salmonid life cycle monitoring station 2013−2014. California Department of Fish and Wildlife, Anadromous Fisheries Resource Assessment and Monitoring Program, Arcata, California.

Ricker, S., and K. Lindke. 2014. California Coastal Salmonid Monitoring Program Annual Report: results of regional spawning ground surveys and estimates of total salmonid redd construction in Mattole River, Humboldt and Mendocino counties California, 2011.

Ricker, S., K. Lindke, and C. Anderson. 2014a. California Coastal Salmonid Monitoring Program Annual Report: results of regional spawning ground surveys and estimates of total salmonid redd construction in the Redwood Creek, Humboldt County California, 2010.

Ricker, S., K. Lindke, and C. Anderson. 2014b. California Coastal Salmonid Monitoring Program Annual Report: results of regional spawning ground surveys and estimates of total salmonid redd construction in the Redwood Creek, Humboldt County California, 2011.

Ricker, S., K. Lindke, and C. Anderson. 2014c. California Coastal Salmonid Monitoring Program Annual Report: results of regional spawning ground surveys and estimates of total salmonid redd construction in the Redwood Creek, Humboldt County California, 2012.

Ricker, S., K. Lindke, and C. Anderson. 2014d. California Coastal Salmonid Monitoring Program Annual Report: results of regional spawning ground surveys and estimates of total salmonid redd construction in the Redwood Creek, Humboldt County California, 2013.

Ricker, S., K. Lindke, and C. Thompson. 2014e. California Coastal Salmonid Monitoring Program Annual Report: results of regional spawning ground surveys and estimates of total salmonid redd construction in Mattole River, Humboldt and Mendocino counties California, 2012.

Ricker, S., M. Groff, and A. Renger. 2015a. California Coastal Salmonid Monitoring Program Annual Report: results of regional spawning ground surveys and estimates of salmonid redd construction in South Fork Eel River, Humboldt and Mendocino counties California, 2010.

Ricker, S., M. Groff, and A. Renger. 2015b. California Coastal Salmonid Monitoring Program Annual Report: results of regional spawning ground surveys and estimates of salmonid redd construction in South Fork Eel River, Humboldt and Mendocino counties California, 2011.

Ricker, S., M. Groff, and A. Renger. 2015c. California Coastal Salmonid Monitoring Program Annual Report: results of regional spawning ground surveys and estimates of salmonid redd construction in South Fork Eel River, Humboldt and Mendocino counties California, 2012.

Ricker, S., M. Groff, and A. Renger. 2015d. California Coastal Salmonid Monitoring Program Annual Report: results of regional spawning ground surveys and estimates of salmonid redd construction in South Fork Eel River, Humboldt and Mendocino counties California, 2013.

Ricker, S., K. Lindke, and C. Anderson. 2015e. California Coastal Salmonid Monitoring Program Annual Report: results of regional spawning ground surveys and estimates of salmonid redd construction in the Humboldt Bay, Humboldt County California, 2010.

Ricker, S., K. Lindke, and C. Anderson. 2015f. California Coastal Salmonid Monitoring Program Annual Report: results of regional spawning ground surveys and estimates of salmonid redd construction in the Humboldt Bay, Humboldt County California, 2011.

Ricker, S., K. Lindke, and C. Anderson. 2015g. California Coastal Salmonid Monitoring Program Annual Report: results of regional spawning ground surveys and estimates of salmonid redd construction in the Humboldt Bay, Humboldt County California, 2012.

Ricker, S., K. Lindke, and C. Anderson. 2015h. California Coastal Salmonid Monitoring Program Annual Report: results of regional spawning ground surveys and estimates of salmonid redd construction in the Humboldt Bay, Humboldt County California, 2013.

RMT (Lower Yuba River Accord River Management Team). 2015. Unpublished data: table of estimated numbers of Chinook salmon, ad-clipped and non ad-cliped phenotypic spring-run Chinook salmon that passed upstream of Daguerre Point Dam annual from 2014 through 2014..

Robinson, T., C. Hanson, S. Engblom, S. Volan, J. Baldrige, L. Riege, B. Wales, A. Shahroody, and C. Lawler. 2009. Summary and analysis of annual fishery monitoring in the lower Santa Ynez River, 1993−2004. Cachuma Conservation Release Board, Santa Barbara, California.

Roegner, G. C., J. A. Needoba, and A. M. Baptista. 2011. Coastal upwelling supplies oxygen-depleted water to the Columbia River estuary. PLoS ONE 6(4):e18672.

Ruiz-Capos, G., and E. P. Pister. 1995. Distribution, habitat, and current status of the San Pedro Mártir rainbow trout, *Oncorhynchus mykiss nelsoni* (Evermann). Bulletin of the Southern California Academy of Sciences 94:131−148.

Rundio, D. E., T. H. Williams, D. E. Pearse, and S. T. Lindley. 2012. Male-biased sex ratio of nonanadromous *Oncorhynchus mykiss* in a partially migratory population in California. Ecology of Freshwater Fish 21:293−299.

Rykaczewski, R. R., J. P. Dunne, W. J. Sydeman, M. García-Reyes, B. A. Black, and S. J. Bograd. 2015. Poleward displacement of coastal upwelling-favorable winds in the ocean's eastern boundary currents through the 21st century. Geophysical Research Letters 42:6424−6431.

SRTTG (Sacramento River Temperature Task Group). 2014. Annual report of activities. 1 October 2013 through 30 September 2014. http://deltacouncil.ca.gov/sites/default/files/2014/10/SRTTG-Annual-Report-10-6-14-final.pdf

SJRGA (San Joaquin River Group Authority). 2011. Technical Report of the San Joaquin River Agreement Vernalis Adaptive Management Plan. http://www.sjrg.org/technicalreport/2011/2011_SJRGA_AnnualTechnicalReport.pdf

Salathé, E. P., A. F. Hamlet, C. F. Mass, S.-Y. Lee, M. Stumbaugh, and R. Steed. 2014. Estimates of twenty-first-century flood risk in the Pacific Northwest based on regional climate model simulations. Journal of Hydrometeorology 15:1881−1899.

Satterthwaite, W. H., M. P. Beakes, E. M. Collins, D. R. Swank, J. E. Merz, R. G. Titus, S. M. Sogard, and M. Mangel. 2009. Steelhead life history on California's central coast: insights from a state-dependent model. Transactions of the American Fisheries Society 138:532−548.

Satterthwaite, W. H., S. A. Hayes, J. E. Merz, S. M. Sogard, D. M. Frechette, and M. Mangel. 2012. State-dependent migration timing and use of multiple habitat types in anadromous salmonids. Transactions of the American Fisheries Society 141:781−794.

Satterthwaite, W.H., M.P. Beakes, E.M. Collins, D.R. Swank, J.E. Merz, R.G. Titus, S.M. Sogard, and M. Mangel. 2010. State-dependent life history models in a changing (and regulated) environment: steelhead in the California Central Valley. Ecological Applications 3:221−243.

Sawaske, S. R., and D. L. Freyberg. 2014. An analysis of trends in baseflow recession and low-flows in rain-dominated coastal streams of the Pacific coast. Journal of Hydrology 519:599−610.

Scheuerell, M. D., R. W. Zabel, and B. P. Sandford. 2009. Relating juvenile migration timing and survival to adulthood in two species of threatened Pacific salmon (*Oncorhynchus* spp.). Journal of Applied Ecology 46:983−990.

Snover, A. K., G. S. Mauger, L. C. W. Binder, M. Krosby, and I. Tohver. 2013. Climate change impacts and adaptation in Washington State: Technical summaries for decision makers. State of Knowledge Report prepared for the Washington State Department of Ecology. Climate Impacts Group, University of Washington, Seattle, Washington.

Snyder, M. A., L. C. Sloan, N. S. Diffenbaugh, and J. L. Bell. 2003. Future climate change and upwelling in the California Current. Geophysical Research Letters 30:1823.

Sounhein, B., E. Brown, M. Lewis, and M. Weeber. 2014. Status of Oregon stocks of coho salmon, 2013. Monitoring Program Report Number OPSW-ODFW-2014-3, Oregon Department of Fish and Wildlife, Salem, Oregon.

Spence, B. C., and J. D. Hall. 2010. Spatiotemporal patterns in migration timing of coho salmon (*Oncorhynchus kisutch*) smolts in North America. Canadian Journal of Fisheries and Aquatic Sciences 67:1316−1334.

Spence, B. C., and T. H. Williams. 2011. Status review update for Pacific salmon and steelhead listed under the Endangered Species Act: Central California Coast coho salmon ESU. U.S. Department of Commerce, NOAA Technical Memorandum NOAA-TM-NMFS-SWFSC-475.

Spence, B. C., E. P. Bjorkstedt, J. C. Garza, J. J. Smith, D. G. Hankin, D. Fuller, W. E. Jones, R. Macedo, T. H. Williams, and E. Mora. 2008. A framework for assessing the viability of threatened and endangered salmon and steelhead in the North-Central California Coast Recovery Domain. U.S. Department of Commerce, NOAA Technical Memorandum NMFS-SWFSC-423.

Spence, B. C., W. G. Duffy, J. C. Garza, B. C. Harvey, S. M. Sogard, L. A. Weitkamp, T. H. Williams, and D. A. Boughton. 2011. Historical occurrence of coho salmon (*Oncorhynchus kisutch*) in streams of the Santa Cruz Mountain region of California: response to an Endangered Species Act petition to delist coho salmon south of San Francisco Bay. U.S. Department of Commerce, NOAA Technical Memorandum NOAA-TM-NMFS-SWFSC-472.

Stachura, M. M., N. J. Mantua, and M. D. Scheuerell. 2014. Oceanographic influences on patterns in North Pacific salmon abundance. Canadian Journal of Fisheries and Aquatic Sciences 71:226−235.

Stillwater-Sciences, R. Dagit, and J. C. Garza. 2010. Lifecycle monitoring of *O. mykiss* in Topanga Creek, California. Prepared for California Department of Fish and Game.

Stout, H. A., P. W. Lawson, D. Bottom, T. Cooney, M. Ford, C. Jordan, R. Kope, L. Kruzic, G. Pess, G. Reeves, M. Sheuerell, T. Wainwright, R. Waples, L. Weitkamp, J. Williams, and T. Williams. 2010. Scientific conclusions of the status review for Oregon Coast coho salmon (*Oncorhynchus kisutch*). Draft report from the Biological Review Team dated May 20, 2010. Northwest Fisheries Science Center, Seattle, Washington.

Sweet, W., J. Park, J. Marra, C. Zervas, and S. Gill. 2014. Sea level rise and nuisance flood frequency changes around the United States. NOAA Technical Report NOS-CO-OPS-073.

Sydeman, W. J., S. A. Thompson, M. García-Reyes, M. Kahru, W. T. Peterson, and J. L. Largier. 2014. Multivariate ocean-climate indicators (MOCI) for the central California Current: Environmental change, 1990−2010. Progress in Oceanography 120:352−369.

Thomas, R. L. 1996. Enhancing threatened salmonid populations: a better way. Fisheries 21:12−14.

Thompson, J. N., and D. A. Beauchamp. 2014. Size-selective mortality of steelhead during freshwater and marine life stages related to freshwater growth in the Skagit River, Washington. Transactions of the American Fisheries Society 143:910−925.

Thompson, S. 2015. Northern Region (R1) Field Note. Van Duzen River, Humboldt County, August 13, 2015 to August 25, 2015 (Summer steelhead surveys in Van Duzen River). California Department of Fish and Wildlife, Fort Bragg, California.

Tohver, I. M., A. F. Hamlet, and S.-Y. Lee. 2014. Impacts of 21st century climate change on hydrologic extremes in the Pacific northwest region of North American. Journal of the American Water Resources Association 50:1461−1476.

USBR and DWR (U.S. Bureau of Reclamation and California Department of Water Resources). 2012. Yolo Bypass salmonid habitat restoration and fish passage implementation plan: Long-term operation of the Central Valley Project and State Water Project Biological Opinion. Reasonable and Prudent Alternative Actions 1.6.1 and 1.7. http://www.water.ca.gov/fishpassage/docs/yolo2.pdf

USFWS (U.S. Fish and Wildlife Service). 2001. Final Restoration Plan for the Anadromous Fish Restoration Program. http://www.fws.gov/sacramento/fisheries/CAMP-Program/Home/Documents/Final_Restoration_Plan_for_the_AFRP.pdf

USFWS (U.S. Fish and Wildlife Service). 2014. Unpublished data: tables of the percentage of Feather River Hatchery coded-wire tagged adults retrieved in the carcass survey on Clear Creek upstream of the weir 2003−2013 and Battle Creek 2010−2013. Red Bluff Fish and Wildlife Office, Red Bluff, California.

USFWS (U.S. Fish and Wildlife Service). 2015. Unpublished data: spreadsheet, total number of adults > 17 inches passing the weir on Battle Creek between 2012−2014. Red Bluff Fish and Wildlife Office, Red Bluff, California.

Wade, A. A., T. J. Beechie, E. Fleishman, N. J. Mantua, H. Wu, J. S. Kimball, D. M. Stoms, and J. A. Stanford. 2013. Steelhead vulnerability to climate change in the Pacific Northwest. Journal of Applied Ecology 50:1093−1104.

Wainwright, T. C., and L. A. Weitkamp. 2013. Effects of climate change on Oregon coast Coho salmon: Habitat and life-cycle interactions. Northwest Science 87:219−242.

Walters, A. W., K. K. Bartz, and M. M. McClure. 2013. Interactive effects of water diversion and climate change for juvenile Chinook salmon in the Lemhi River Basin (U.S.A.). Conservation Biology 27:1179−1189.

Waples, R. S. 1991. Pacific salmon, *Oncorhynchus* spp., and the definition of "species" under the Endangered Species Act. Marine Fisheries Review 53:11−22.

Ward, B. R. 2000. Declivity in steelhead (*Oncorhynchus mykiss*) recruitment at the Keogh River over the past decade. Canadian Journal of Fisheries and Aquatic Sciences 57:298-306.

Watry, C., A. Gray, K. Jones, K. Sellheim, and J. Merz. 2012. Juvenile salmonid out-migration at Caswell Memorial Park in the Lower Stanislaus River, California 2010−11 Biannual Report. Prepared for the USFWS Comprehensive Assessment and Monitoring Program, Grant No. 813326G008.

Weitkamp, L. A. 2010. Marine distributions of Chinook salmon from the West Coast of North America determined by coded wire tag recoveries. Transactions of the American Fisheries Society 139:147−170.

Weitkamp, L. A., T. C. Wainwright, G. J. Bryant, G. B. Milner, D. J. Teel, R. G. Kope, and R. S. Waples. 1995. Status review of coho salmon from Washington, Oregon, and California. U.S. Department of Commerce, NOAA Technical Memorandum NMFS-NWFSC-24.

Williams, A. P., R. Seager, J. T. Abatzoglou, B. I. Cook, J. E. Smerdon, and E. R. Cook. 2015. Contribution of anthropogenic warming to California drought during 2012–2014. Geophysical Research Letters 42:6819−6828.

Williams, D. 2015. Yurok tribal fisheries harvest of species listed under the Endangered Species Act, 2014. Yurok Tribal Fisheries Program, Klamath, California.

Williams, T. H., E. P. Bjorkstedt, W. Duffy, D. Hillemeier, G. Kautsky, T. Lisle, M. McCain, M. Rode, R. G. Szerlong, R. Schick, M. Goslin, and A. Agrawal. 2006. Historical population structure of coho salmon in the Southern Oregon/Northern California Coasts Evolutionarily Significant Unit. U.S. Department Commerce, NOAA Technical Memorandum NMFS-SWFSC-390.

Williams, T. H., S. T. Lindley, B. C. Spence, and D. A. Boughton. 2011. Status review update for Pacific salmon and steelhead listed under the Endangered Species Act: Southwest. National Marine Fisheries Service, Southwest Fisheries Science Center, Santa Cruz, California.

Williams, T. H., B. C. Spence, W. Duffy, D. Hillemeier, G. Kautsky, T. E. Lisle, M. McCain, T. E. Nickelson, E. Mora, and T. Pearson. 2008. Framework for assessing viability of threatened coho in the Southern Oregon/Northern California Coast Evolutionarily Significant Unit. U.S. Department of Commerce, NOAA Technical Memorandum NOAA-TM-NMFS-SWFSC-432.

Wing, B. L. 2006. Unusual invertebrates and fish observed in the Gulf of Alaska, 2004−2005. PICES Press 14:26−28.

Workman, M. 2003. Lower Mokelumne River upstream fish migration monitoring conducted at Woodbridge Irrigation District Dam, August 2002 through July 2003. East Bay Municipal Utility District, Lodi, California.

Wright, S. 1931. Evolution in Mendelian populations. Genetics 16:97−159.

Yang, Z., T. Wang, N. Voisin, and A. Copping. 2015. Estuarine response to river flow and sea-level rise under future climate change and human development. Estuarine, Coastal and Shelf Science 156:19−30.

Zabel, R. W., M. D. Scheuerell, M. M. McClure, and J. G. Williams. 2006. The interplay between climate variability and density dependence in the population viability of Chinook salmon. Conservation Biology 20:190−200.

Zimmerman, C. E., G. W. Edwards, K. Perry. 2009. Maternal origin and migratory history of steelhead and rainbow trout captured in rivers of the Central Valley, California. Transactions of the American Fisheries Society 138:280–291.

Appendix A. Sources of data used in assessment of status of populations of salmon and steelhead in the NCCC Recovery Domain. Minor sources used to establish species occurrence are not included.

Population/Watershed	CCC-Coho salmon	CC-Chinook salmon	NC-Steelhead	CCC-Steelhead	Sources
Redwood Creek (Humboldt)		X	X (winter)		Ricker et al. 2014a,b,c,d
			X (summer)		Anderson 2015 (unpublished data)
Prairie Creek		X	X (winter)		Duffy 2013 (unpublished data)
Mad River					
Cannon Creek		X			PFMC 2015a
Humboldt Bay		X	X		Ricker et al. 2015e,f,g,h
Freshwater Creek		X	X (winter)		Ricker 2015
Eel River					
S. FK. Eel River		X	X		Ricker et al. 2015a,b,c,d
Van Duzen River			X (summer)		Thompson 2015
Sproul Creek		X			PFMC 2015a
Tomki Creek		X			PFMC 2015a
Van Arsdale Station		X	X (winter)		Williams et al. 2011; Harris and Thompson 2013, 2014; Harris CDFW (personal communication)*
Middle Fork Eel River			X (summer)		Harris and Thompson 2014; S. Harris, CDFW (personal communication)*
Mattole River		X	X (winter)		Ricker and Lindke 2014; Ricker et al. 2014e
			X (summer)		MSG 2015
Usal Creek	X		X		Holloway et al. 2014 (draft); Gallagher et al. 2014
Cottaneva Creek	X		X		Holloway et al. 2014 (draft); Gallagher et al. 2014
Wages Creek	X		X		Holloway et al. 2014 (draft); Gallagher et al. 2014
Ten Mile River	X	X	X		Holloway et al. 2014 (draft); Gallagher et al. 2014
Pudding Creek	X		X		Holloway et al. 2014 (draft); Gallagher et al. 2014
Noyo River	X	X	X		Holloway et al. 2014 (draft); Gallagher et al. 2014
S. Fk. Noyo River	X		X		Holloway et al. 2014 (draft); Gallagher et al. 2014
Hare Creek	X		X		Holloway et al. 2014 (draft); Gallagher et al. 2014
Caspar Creek	X		X		Holloway et al. 2014 (draft); Gallagher et al. 2014
Big River	X	X	X		Holloway et al. 2014 (draft); Gallagher et al. 2014
Little River	X		X		Holloway et al. 2014 (draft); Gallagher et al. 2014
Albion River	X	X	X		Holloway et al. 2014 (draft); Gallagher et al. 2014

Appendix A. continued.

Population/Watershed	CCC-Coho salmon	CC-Chinook salmon	NC-Steelhead	CCC-Steelhead	Sources
Big Salmon Creek	X		X		Holloway et al. 2014 (draft); Gallagher et al. 2014
Navarro River	X	X	X		Holloway et al. 2014 (draft); Gallagher et al. 2014
N. Fk. Navarro River	X		X		Holloway et al. 2014 (draft); Gallagher et al. 2014
Greenwood Creek	X		X		Holloway et al. 2014 (draft); Gallagher et al. 2014
Elk Creek	X		X		Holloway et al. 2014 (draft); Gallagher et al. 2014
Brush Creek	X		X		Holloway et al. 2014 (draft); Gallagher et al. 2014
Garcia River	X	X	X		Holloway et al. 2014 (draft); Gallagher et al. 2014
Gualala River					
Wheatfield Fk Gualala R.			X		DeHaven 2010
Russian River		X			Horton 2015
Lagunitas Creek	X				Ettlinger et al. 2015
Pine Gulch	X			X	Carlisle and Reichmuth 2015; M. Reichmuth (unpublished data)
Redwood Creek (Marin)	X			X	Carlisle and Reichmuth 2015; Reichmuth 2015
San Pedro Creek				X	Jankovitz 2013†
San Gregorio Creek				X	Goin 2014; Goin 2015
Pescadero Creek	X			X	Jankovitz 2013†; Goin 2014; Goin 2015
Gazos Creek	X			X	Jankovitz 2013†; Goin 2014; Goin 2015
Waddell Creek	X			X	Jankovitz 2013†; Goin 2014
Scott Creek	X			X	Kiernan 2015 (unpublished data)
San Vicente Creek	X			X	Jankovitz 2013†; Goin 2014; Goin 2015
San Lorenzo River	X			X	Jankovitz 2013†; Goin 2014; Goin 2015
Soquel Creek				X	Goin 2015
Aptos Creek	X			X	Jankovitz 2013†

* − Unpublished data for Van Arsdale Station and the Middle Fork Eel River for 2014 were provided to NMFS by S. Harris, CDFW. Data from Van Arsdale Station prior to 2011 can be found in annual administrative reports produced by California Department of Fish and Wildlife; these data were summarized graphically in Williams et al. 2011.

† − Report undated but presumed to be 2013.